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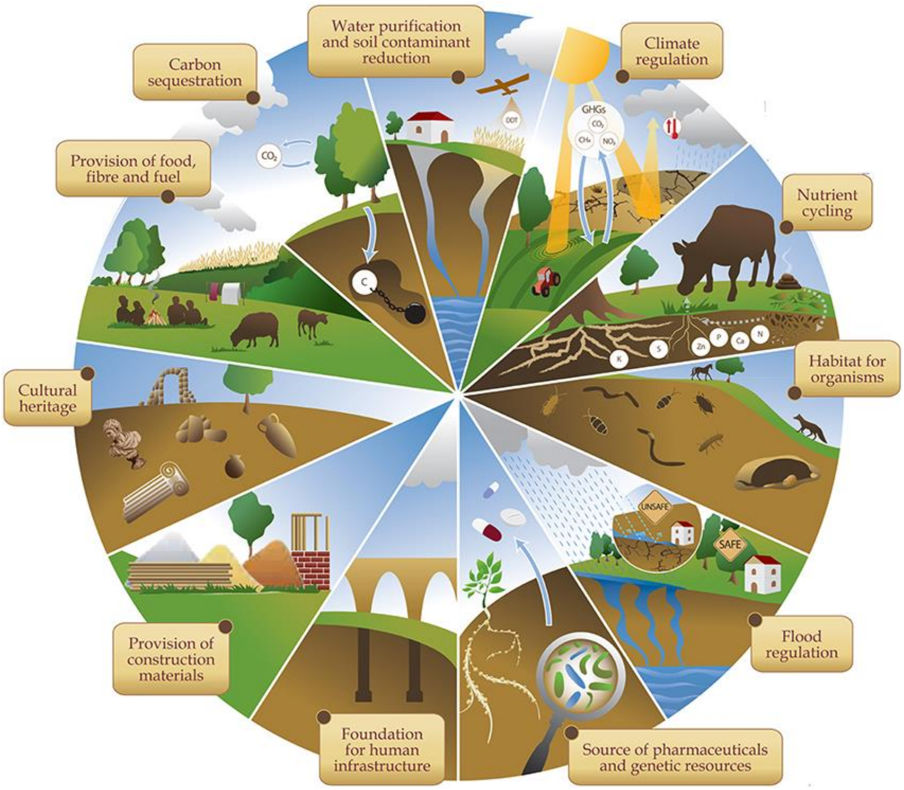

Soil Functions and Ecosystem Services

A Literature Review (Part 2/2)

PAUL DRENNING

Department of Architecture and Civil Engineering
Division of Geology and Geotechnics
Research Group Engineering Geology
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Gothenburg, 2021

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Cover: Schematic diagram of soil functions from the FAO, from Baveye et al. (2016), CC-BY 4.0.

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SUMMARY

Soils are a non-renewable resource and comprise a key component of the world's stock of natural capital. Due to industrialisation, urbanisation and other patterns of unsustainable development, widespread land degradation in the form of contamination, soil sealing, compaction, etc. has impaired the capacity of soils to perform their essential functions and provide humans with vital ecosystem services. Brownfields are typically urban or peri-urban sites that have been affected by the former uses of the site, are or are perceived to be contaminated, and require intervention to bring them back to beneficial use. They also constitute an important and underutilised land and soil resource to provide ecosystem services in urban areas as an element of green infrastructure through the use of nature-based solutions such as gentle remediation options (GRO). Within the scope of the Ph.D. project " Enhancing ecosystem services by innovative remediation using gentle remediation options (ECO-GRO)", an in-depth but inexhaustive literature review has been carried out to build a theoretical understanding of soil functions and ecosystem services for the overall research project. This literature review report (part 2 of 2) will present a compilation of the main findings by continuing with E) core concepts of soil biology, functioning and linkages to ecosystem services including how they can be delivered in healthy soils by functional assemblages of soil biota; then, F) methods for assessing soil quality are reviewed including potential physical, chemical and biological indicators, how they can be selected using a logical sieve approach, which standardised analyses exist to measure certain parameters as well as how they can be interpreted to give an indication of the status of certain aspects of soil functioning; G) quantitative, semi-quantitative and qualitative methods for assessing ecosystem services are also discussed, primarily within the context of urban or brownfield soils, and noteworthy examples are presented at some length as well as considerations for economic valuation of ecosystem services; and finally H) broader implications for land management and planning are considered in terms of managing soils to improve their quality and adaptive management and monitoring approaches to iteratively evaluate soils for their capacity to function and deliver ecosystem services over time. Also, how the breadth of information presented in this report can be transferred and applied at contaminated sites and marginal land to improve soil quality and provide much-needed ecosystem services, particularly in urban areas, is discussed.

Key words:

Ecosystem services (ES), soil functions (SF), soil quality indicators (SQI), soil quality assessment, ecosystem service assessment, brownfields, contaminated sites, sustainable remediation, ecological risk assessment (ERA), gentle remediation options (GRO)

Preface

This literature review is the 2nd part of two literature reviews which has been carried out at the Department of Architecture and Civil Engineering, Division of Geology and Geotechnics at Chalmers University of Technology in Gothenburg, Sweden. The work has been supervised by Docent Jenny Norrman and is performed as part of a Ph.D. research project entitled "Enhancing ecosystem services by innovative remediation using gentle remediation options (ECO-GRO)" which has been funded by the Swedish research council FORMAS (Grant number: 2018-01467)

Göteborg, Sweden

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Paul Drenning

List of notations

Abbreviations

ARISA	Automated rRNA intergenic spacer analysis
CBA	Cost –benefit analysis
DST	Decision support tool
ERA	Ecological (or ecosystem) risk assessment
ES	Ecosystem services
MBC	Microbial biomass carbon
MCA	Multi-criteria analysis
MCDA	Multi-criteria decision analysis
MDS	Minimum dataset
qPCR	Quantitative polymerase chain reaction
SF	Soil functions
SDG	Sustainable Development Goals
SIR	Substrate-induced respiration
SQI	Soil quality indicators
TRFLP	Terminal restriction length polymorphism

Contents

1 Introduction

1.1 Background

Natural capital is defined by the Natural Capital Forum as the world's stock of natural assets including geology, soil, air, water and all living things (www.naturalcapitalforum.com). According to the European Commission, a major problem connected to our current resource consumption patterns is that our common pools of natural capital are treated as infinite, 'free' commodities whose value is not sufficiently accounted for in modern economic markets (EC, 2011a). This has inevitably led to detrimental resource depletion, pollution, and a wide range of associated threats to our long-term sustainability and resilience to environmental shocks, especially in urban areas (EC, 2011a; Olofsdotter et al., 2013). At present, resource consumption in urban areas accounts for almost 80% of global emissions of greenhouse gases. The legacy of industrialization over the past century has added the problem of widespread contamination in and around cities' soil and water systems. And cities in general are pushing at the limits of the established planetary boundaries (Olofsdotter et al., 2013; Rockström et al., 2009; Steffen et al., 2015). A series of agenda-setting reports by the European commission (e.g. *Vision for a Resource Efficient Europe, European Biodiversity Strategy to 2020*) have raised awareness of the widespread degradation of ecosystems by over-exploitation, land-use change, contamination, sealing, compaction, erosion, neglect, etc. which have led to rapid losses in biodiversity and diminished the total provided ecosystem services by approximately 60% worldwide in the past 50 years alone (EC, 2011a, 2011b, 2006; Ellen MacArthur Foundation, 2015). The concept of ecosystem services (ES) has become increasingly prevalent to denote nature's contribution to human welfare, and is commonly defined as 'the goods and services that humans derive from natural and human-modified systems on which societal welfare and economic development directly depend' (Millennium Ecosystem Assessment, 2005; TEEB, 2010). Soil as well can be considered a non-renewable resource as it takes many hundreds of years to form fertile topsoil, and land itself is a finite and shrinking resource (Breure et al., 2018). From an anthropocentric point-of-view, protecting and restoring these natural assets is imperative to human well-being for current and future generations. Urgent action is mandated by the European Commission and United Nations to curb the loss of biodiversity, resource degradation, and land-take by transitioning to a more sustainable development pattern that protects and preserves the value that these ecosystems represent.

Given the situation, soil and its functions have been raised to a position of critical importance for our common future through the (currently revisited) *Thematic Strategy on Soil Protection* (EC, 2006). Within the *Thematic Strategy*, seven essential soil functions (SF) have been established: (i) biomass production, including agriculture and forestry; (ii) storing, filtering and transforming nutrients, substances and water; (iii) biodiversity pool, such as habitats, species and genes: (iv) physical and cultural environment for humans and human activities; (v) source of raw materials; (vi) acting as a carbon pool; (vii) archive of geological and archaeological heritage (EC, 2006). The significance of SF and soil-based ecosystem services (ES) for realising the UN's Sustainable Development Goals (SDG) has also been addressed by directly linking them to many of the SDGs (e.g. S. Keesstra et al. 2018; S. D. Keesstra et al. 2016). Soil functions, Figure 1-1, are critical for the delivery of ecosystem services to humans, and thus it is critical to account for these and evaluate soil performance in urban development to maximize soil multi-functionality and SF and ES provisioning whenever possible (Bünemann et al., 2018; Lehmann and Stahr, 2010; Volchko et al., 2013, 2014a, 2019).

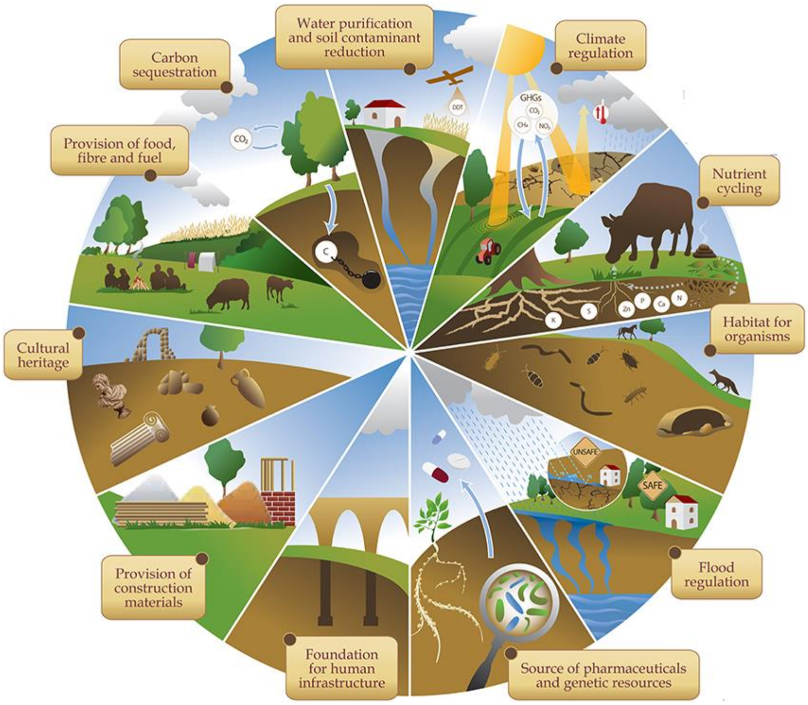


Figure 1-1. Schematic diagram of soil functions from the FAO, from (Baveye et al., 2016) *(CC-BY 4.0).*

Brownfields are underused areas with, in many cases, real or perceived soil and groundwater contamination which often is a barrier to redevelopment in terms of investment risks, ownership constraints, risk of future liability claims and public stigma (Ferber et al., 2006; Norrman et al., 2016). In Europe, there are more than 2.5 million potentially contaminated sites caused by anthropogenic activity, i.e. brownfields, of which approximately 85 000 are in Sweden (Panagos et al., 2013). Soil contamination, along with other degradation processes, can negatively affect soil health (FAO et al., 2020; FAO and UNEP, 2021; Orgiazzi et al., 2016; Turbé et al., 2010), which is defined as '*the capacity of a given soil to perform its functions as a living system capable of sustaining biological productivity, promoting environmental quality and maintaining plant and animal health* (Doran and Zeiss, 2000). Instead of being viewed as a valuable resource to be cleaned and reused, contaminated soil is often viewed as a disposable waste, so conventional "quick and dirty" remediation techniques, usually based on removing or destroying the source of contamination, tend to entail irreversible damage to ecosystems (FAO et al., 2020; Gerhardt et al., 2017; Mench et al., 2010). Conventional remediation techniques are often resource intensive and entail multiple environmental externalities often resulting in a lifeless soil ecosystem unfit for 'soft' end uses like green spaces which require ecological functioning (Bardos et al., 2016; FAO et al., 2020; G. Lacalle et al., 2020; Volchko et al., 2014a). New practices are crucial for sustainable remediation and brownfield regeneration, because a significant amount of brownfield land area remains derelict or underutilized due to restoration being uneconomic or unsustainable using conventional methods (Bardos, 2014; Bardos et al., 2020, 2016, 2018). This problem is of particular concern for large land areas or smaller, marginal sites where contamination inhibits immediate development, but economic return post-remediation does not justify the costs (Cundy et al., 2016). A promising field of innovative remediation technologies which have received much attention in recent years are those involving plant- (phyto-), fungi- (myco-) and/or bacteria- (bio-) based methods with or without the use of soil amendments, i.e. gentle remediation options (GRO). Research shows that GRO can provide both effective risk management and a net gain in ecological soil function (Cundy et al., 2016; Mench et al., 2010; Vangronsveld et al., 2009).

1.2 Aim and Scope

A literature review has been carried out as part of the Ph.D.-project "Enhancing ecosystem services by innovative remediation using gentle remediation options (ECO-GRO)". This report presents an in-depth but inexhaustive compilation of information that will be used to build a theoretical foundation for the overall research project. Concepts covered include soil biology, soil functioning, ecosystem functioning, and other fundamental soil science related concepts, which are referred to specifically in relevant sections. This report also aims to give a brief review of methods for soil quality and ecosystem service assessment including possible economic valuation of ecosystem services and implications for land and soil management.

Given the growing importance of soils for achieving environmental goals and their recognition as a non-renewable resource, an in-depth understanding of the soil system is crucial for discussing the improvement of provisioning of soil functions and ecosystem services, including at contaminated sites. A thorough assessment of what this involves lies outside the scope this review, which instead will focus on synthesizing the state-of-the-art and presenting need-to-know concepts within the context of the Ph.D.-project. This report is the 2nd part of two literature reviews concerning: 1) Gentle remediation options (GRO) and 2) Soil functions and ecosystem services. GRO are the focus of the 1st part of literature review and will not be discussed in-depth in this report.

Specific objectives with this literature review:

- Target literature review towards practical information necessary to carry out pilot studies, i.e. *what do we need to know to assess soil functions and ecosystem services?*
- Compile pertinent studies to create a reference bank for assessing soil functions and ecosystem services in different situations.
- Identify influential sources or seminal works that lead the field to focus on for deriving the most valuable information.
- Gain the necessary background knowledge to have a sufficient understanding of the topics or themes addressed in the ECO-GRO Ph.D.-project and to identify areas of prior research to prevent duplication of effort (i.e. not re-inventing the wheel).
- Identify key themes and the intersectionality between related (yet disconnected) fields: gentle remediation options to soil functioning and ecosystem service provisioning. This will be accomplished within the ECO-GRO Ph.D.-project by combining the findings of both parts of literature review in the future thesis work.

1.3 Methodology

The overarching purpose of a literature review can be broadly described as a more or less systematic way of collecting and synthesizing previous research (Snyder, 2019). In addition to a number of seminal works and highly cited and relevant papers (referred to explicitly in relevant sections), a series of steps (according to the Chalmers Library Literature Review Guide[1]) were followed to create a 'semi-systematic' method to add a robustness to the review in searching for supplementary material, including:

1. **Problem formulation** – establishing the thematic areas and topics to be covered in the review, broadly including:

[1] Literature Review Guide at Chalmers University of Technology

- Interactions between soil biota and the environment that underly soil functions and the delivery of ecosystem services.
- Methods and indicators to use in soil quality assessment to monitor and manage soil biodiversity for the delivery of ecosystem services.
- Identification of existing ecosystem services at site and methods to inventory them, including via ecological risk assessment.
- Identification of current and future needs of ecosystem services.
- Effects on ecosystem services from disturbances and management practices.
- Connections to holistic land management and planning.
- Opportunities and synergies to enhance ecosystem services and improve overall soil quality and functioning, for example by using gentle remediation options.

2. **Formulating sensitive search terms** – including relevant research and exclude the greater bulk, to search for literature in the Scopus database that may be relevant to include in the review.
3. **Screen the selected literature** – by reading titles, abstracts, summaries, etc. to determine which are the most useful to include. Also, identify previously conducted reviews relevant to soil functions and ecosystem services and establish prominent, seminal works that have an outweighed influence in the field to rely more heavily upon.
4. **Analyse and interpret** – analysing the findings and conclusions of the most significant literature, extract the pertinent information, and synthesise according to pre-selected themes to clearly structure within the overall review.

Many searches in the Scopus database were carried out to isolate the most relevant scientific articles to include in this review, shown in Appendix I. To ensure that the searches were performed in a systematic, transparent way the PRISMA method was adopted[2].

1.4 Terminology

Integrating key concepts of soil science like soil quality and soil quality indicators (including ecological soil health) into contaminated site investigation and management is a significant step in the right direction towards sustainable soil and land management where soil is *managed in accordance with the soil's capability and condition* (Volchko et al., 2019). By accounting for soil parameters beyond just contamination levels in decision-making the latent potential of the soil can be leveraged to turn sustainable ambitions to recover ecosystem functions through soil protection into action (Volchko et al., 2019). For, **the ultimate objective of any remediation process must be not only to remove the contaminants from the soils (or instead disrupt the source-pathway-receptor linkages) but also to restore soil quality** (Epelde et al., 2008; FAO et al., 2020; Gómez-Sagasti et al., 2012).

Terms and definitions related to soils can vary significantly and be used interchangeably even within disciplines. To avoid confusion, it is necessary to establish a common language and

[2] http://www.prisma-statement.org/

terminology in the context of contaminated sites for the purposes of this review. Terms will be used according to the following definitions:

Brownfield has been defined a site that has been affected by the former uses of the site or surrounding land, is derelict or underused, is mainly in fully or partly developed urban areas, requires intervention to bring it back into beneficial use, and may have real or perceived contamination problems (Ferber et al., 2006; ISO, 2017a).

Brownfield regeneration/restoration is the management, rehabilitation and return to beneficial use of the brownfield land resource base in such a manner as to ensure the attainment and continued satisfaction of human needs for present and future generations in environmentally non-degrading, economically viable, institutionally robust and socially acceptable ways" (Bardos et al., 2016).

Ecosystem services (ES) are commonly defined as the goods and services that humans derive from natural and human-modified systems on which societal welfare and economic development directly depend (Millennium Ecosystem Assessment, 2005; TEEB, 2010). They are typically divided into 4 categories: i) provisioning (products obtained from ecosystems), ii) regulating (benefits obtained from regulation of ecosystem processes), iii) cultural (non-material benefits obtained from ecosystems), and iv) supporting (services necessary for the production of all other ecosystem services) (Millennium Ecosystem Assessment, 2005; TEEB, 2010).

Gentle remediation options (GRO) are risk management strategies or technologies that result in a net gain (or at least no gross reduction) in soil function as well as achieving effective risk management(Cundy et al., 2016)**.**

Green infrastructure refers to a strategically planned network of natural and seminatural areas with other environmental features designed and managed to deliver a wide range of ecosystem services" (EC, 2013). A similar concept, **blue-green infrastructure** is defined as *interconnected networks of land and water that support species, maintain ecological processes, sustain air and water resources, and contribute to the health and quality of life for communities and people* (Olofsdotter et al., 2013).

Human health is often considered as a basic human right and is defined by the World Health Organization (WHO) as not simply being free from illness, but in a state of complete physical, mental and social well-being. Biodiversity can be considered as the foundation for human health as it underpins the functioning of the ecosystems on which we depend for our food and fresh water; aids in regulating climate, floods and disease; provides recreational benefits and offers aesthetic and spiritual enrichment. Biodiversity also contributes to local livelihoods, to both traditional and modern medicines and to economic development (Health and Biodiversity (cbd.int)).

Natural capital refers to the extension of the economic idea of manufactured capital to include environmental goods and services (Dominati et al., 2010). Natural capital consists of *stocks of natural assets (e.g. soils, forests, water bodies) that yield a flow of valuable ecosystem goods or services into the future* (COSTANZA and DALY, 1992; Dominati et al., 2010). Soils are considered here as natural capital and provide services such as recycling of wastes or flood mitigation (Dominati et al., 2010).

Soil biodiversity comprises the variation in soil life, from genes to communities, and the ecological complexes of which they are part, that is from soil microhabitats to landscapes (Turbé et al., 2010). This variation is generally described in terms of three interrelated attributes of biodiversity: composition, structure and function (Pulleman et al., 2012). Biodiversity is then considered as the quantity, variety and structure of all forms of life in soils, as well as related functions (Pulleman et al., 2012).

Soil fertility has its origins in agriculture primarily referring to the ability of the soil to supply essential plant nutrients and soil water in adequate amounts and proportions for plant growth and reproduction in the absence of toxic substances which may inhibit plant growth (Bünemann et al., 2018). Soil fertility is a difficult term for it can be referred to as both soil function and ecosystem service. Whenever possible, this term will be avoided in favour of more consistently used terms like primary productivity.

Soil functions is a loaded term which has been used alternatively to mean process, function, role, or service (Baveye et al., 2016; Bünemann et al., 2018). Confusing as the term may be, it has served as a conceptual foundation in soil management, most notably in EC 2006, so it is considered worthwhile to clarify and distinguish between soil *processes, functions and services* (Baveye et al., 2016). Accordingly, *soil functions* are here defined as *what the soil has the capability to do in its natural (undisturbed) state as a result of the (bundles of) soil processes (*e.g. soil formation, nutrient cycling, etc.*) arising out of the complex interaction between biotic and abiotic components in the soil environment* (Bünemann et al., 2018; Volchko et al., 2013). Soil functions thus can be viewed as *a subset of wider ecosystem functions* (Volchko et al., 2013), *which underpin the delivery of ecosystem services* (Bünemann et al., 2018).

Note: this term is often used interchangeably with *ecosystem functions*.

Soil health accounts for soil's capacity beyond the direct utilitarian end use considerations as it has typically included soil's ecological attributes associated with soil biota, biodiversity, and the living and dynamic nature of soil (Bünemann et al., 2018; Doran and Zeiss, 2000; Garbisu et al., 2011; Karlen et al., 1997). The most frequently referred to definition defines soil health as *the capacity of soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and promote plant and animal health* (Doran and Zeiss, 2000). In a more agricultural context, Kibblewhite et al. (Kibblewhite et al., 2008) derive the definition of soil health as an essential feature of sustainable agriculture: *a healthy agricultural soil is one that is capable of supporting the production of food and fibre, to a level and with a quality sufficient to meet human requirements, together with continued delivery of other ecosystem services that are essential for maintenance of the quality of life for humans and the conservation of biodiversity.* A recently performed review (Bünemann et al., 2018) concluded that soil quality and soil health are essentially equivalent, so for this review the term soil quality will be favoured.

Soil quality has a generally agreed upon definition broadly meaning *the capacity of a soil to perform its functions necessary for its intended end use* (Garbisu et al., 2011; Karlen et al., 2003, 1997; USDA Natural Resource Conservation Service, 2015; Volchko et al., 2013). This inherently anthropocentric definition has been expanded in Bünemann et al. (2018) to more broadly include ecological (i.e. biological) functioning *'within ecosystem and land-use boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health.*' This expanded definition includes soil health (see above) and reflects more the complexity and site-specificity of soil functioning as well as indicates the multi-functionality of soils when functioning according to their capacity.

Soil quality indicators (SQI) have been loosely defined as *'those soil properties and processes that have greatest sensitivity to changes in soil function*' (Andrews et al., 2004), or in an expanded form as *'measurable properties of the soil used to evaluate the degree to which the soil quality matches the soil functions determined by the intended end use of the soil*' (Volchko et al., 2013). SQI may encompass physical, chemical, and biological parameters which can be connected to dynamic soil properties that can be strongly influenced by management and interventions at a site and correlate well with ecosystem processes (Andrews et al., 2004; Bünemann et al., 2018; Volchko et al., 2013).

Soil services can essentially be viewed as soil-based ecosystem services. That is, soil functions which have been utilized by humans directly or indirectly; therefore, are considered soil services (Volchko et al., 2013). *Soil service indicators* have also been integrated into multi-criteria decision analysis (MCDA) frameworks (Volchko et al., 2013, 2014a) as a means to evaluate their contribution to human well-being; however, that lies outside of the scope of this review.

Sustainable remediation is the practice of demonstrating, in terms of environmental, economic and social indicators, that the benefit of undertaking remediation is greater than its impact and that the optimum remediation solution is selected through the use of a balanced decision-making process (Bardos, 2014), or simply the elimination and/or control of unacceptable risks in a safe and timely manner whilst optimising the environmental, social and economic value of the work (ISO, 2017a).

1.5 Structure of the report and the limitations

The literature review report has been structured to present concepts related to soil biology and functioning on micro-scale then broaden to assessment and their broader implications for land management and planning. *Chapter 1* provides and introduction and background to the topic and its relevance to the broader Ph.D. project. *Chapter 2* presents and discusses core concepts related to soil biology, corresponding functions and their linkages to ecosystem services. *Chapter 3* presents the state-of-the-art for soil quality assessment including selection and use of soil quality indicators. *Chapter 4* considers the use of similar indicators and other methods to assess ecosystem services with a focus on brownfields or contaminated sites. *Chapter 5* discusses the broader implications of the covered concepts for land management, planning and monitoring particularly for contaminated sites and marginal land. *Chapter 6* provides a final discussion and concluding remarks.

The main limitation of this literature review is that it is not systematic and wholly inclusive of the field of scientific literature. Instead, a more targeted approach has been taken to focus on the narrower range of topics presented here and by relying primarily on select, highly cited or relevant sources. Also, the deliberate focus of assessing soil functions and ecosystem services at brownfield sites to the exclusion of broader agricultural soil considerations. For this review, the typology of relevant ES will pertain primarily to soil-based services within an urban or brownfield context. The non-anthropocentric, intrinsic value of soils will be addressed in terms of its protection value as a haven for biodiversity for its own sake (e.g. precautionary principle), as it is essential for the health and functioning of ecosystems which can provide ES to humans, but the utilitarian value of soils will be prioritised.

2 Soil biology, functioning and ecosystem services

Soils make up a crucial part of the Earth's system and play fundamental roles in its functioning, upon which humans are dependent, as well as linking the atmosphere, the subsurface, and the aquatic realms (Barrios, 2007; Faber et al., 2013; Kibblewhite et al., 2008). This section will discuss some of the essential aspects of biodiversity driving the ecological functioning of soils, how they can be grouped into understandable and measurable entities, which ecosystem services can be attributed to them and the threats posed to soils that can degrade overall soil functioning. The field of soil biology, function and ecosystem services is vast and many concepts will be covered only superficially; for more information the reader is referred to other extensive, in-depth reports e.g. (FAO et al., 2020; Orgiazzi et al., 2016; Turbé et al., 2010).

2.1 Soil biota

When referring to the soil system, it is common to emphasize the physical or material geochemistry (i.e. abiotic) component and neglect the living organisms (i.e. biotic) that are ultimately responsible for the majority of soil processes (Creamer et al., 2016a; Doran and Zeiss, 2000; Griffiths et al., 2016; Kibblewhite et al., 2008; Ritz et al., 2009). Ritz et al. (2009) state that the physical (e.g. texture, bulk density, porosity, and water availability) and chemical (e.g. pH, organic matter content, metal availability) properties of soils provide the fundamental context, and sets the limits, in which the biotic assemblages operate and thus have a clear utility in assessing ecological status. However, the majority of soil processes are in fact driven by the soil biota, shown grouped according to size in Figure 2-1 and within the context of the soil food web in Figure 2-2. According to Kibblewhite, Ritz, and Swift (2008) the unique and crucial feature of the biota is that it is *adaptive to changes in environmental circumstances, driven by processes of natural selection*, in ways that the abiotic systems of the soil are not.

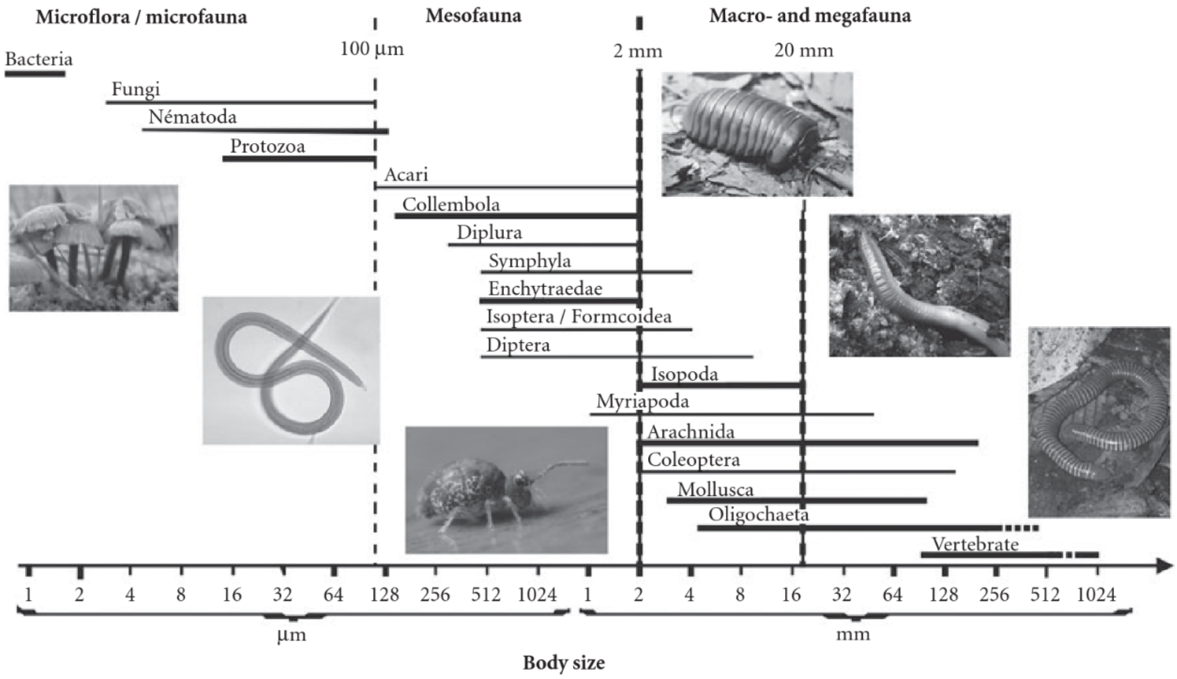


Figure 2-1. Size distribution of soil organisms, from (Lavelle, 2013).

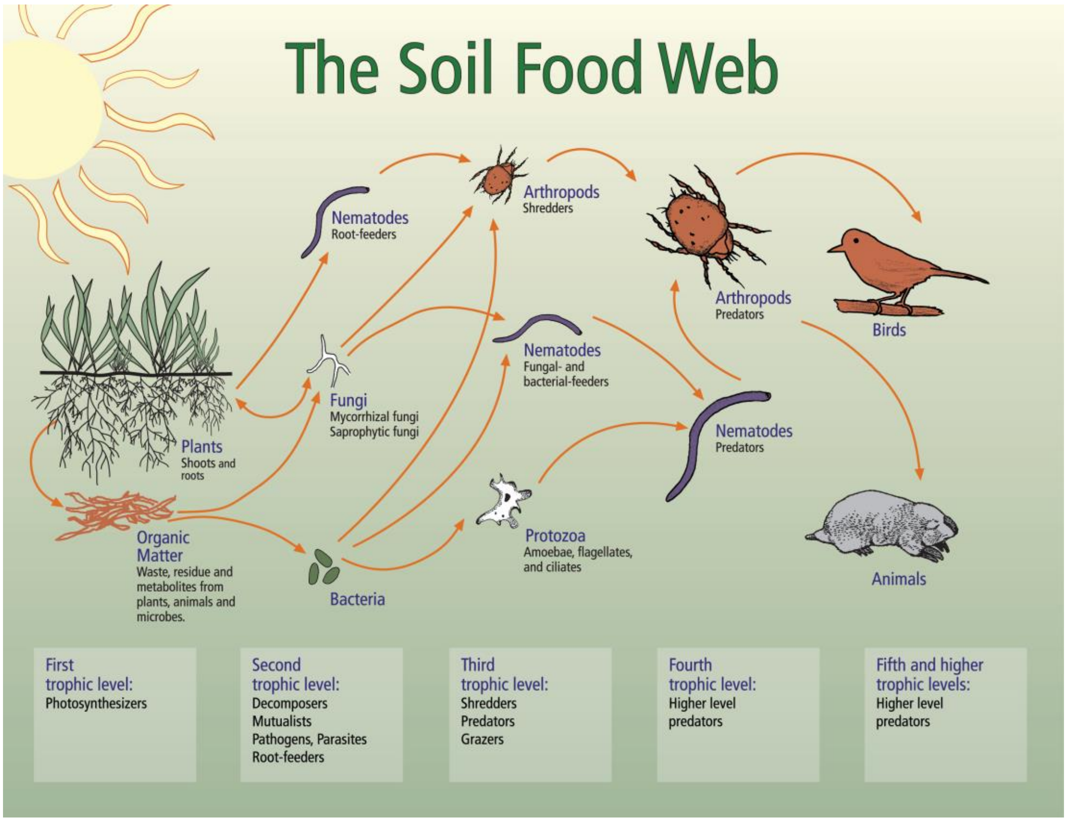


Figure 2-2. The soil food web, from USDA Natural Resources Conservation Service – Soil and Water Conservation Society. Available at: Soil Food Web | NRCS Soils (usda.gov).

As shown in Figure 2-1, soil biota can be broadly separated by size into the following groupings: microbes/microflora, microfauna, mesofauna, macrofauna and megafauna. Which organisms that can be included in these broad groupings and their associated roles in the soil system are briefly discussed in Table 2-1 below:

Table 2-1. Soil biota organised by size class, summarised from (Wurst et al., 2013). *Predominant organisms per group are in bold.*

Size class	Dominating organisms	Soil processes	Associated functions and services
Microbes/ microflora	**Bacteria**, **fungi**, archaea	Degradation of organic matter, nitrogen fixation and denitrification, soil aggregation	Decomposition, carbon and nutrient cycling, disease suppression, regulation of plant growth and primary productivity
Microfauna	**Nematodes**, protozoa	Predation, herbivory, bacteriovory, fungivory parasitism, provide food source to other organisms, distribute microbes in rhizosphere	Nutrient cycling, regulation of population sizes, pest and disease suppression
Mesofauna	**Mites**, **collembola** (springtails), enchytraeids (potworms)	Herbivory, bacteriovory, fungivory, predation, provide food source to other organisms, distribute microbes in rhizosphere	Nutrient cycling, regulation of population sizes, pest and disease suppression
Macrofauna/ megafauna	**Earthworms**, ants, termites, spiders, millipedes, beetles, moles	Degradation of organic matter, predation, herbivory, parasitism, burrowing, soil mixing, soil aggregation, provide food source to other organisms	Decomposition, carbon and nutrient cycling, water regulation, pest and disease suppression, regulation of population sizes, positive/negative effects on plant growth and primary productivity

Wurst et al. (2013) demonstrate the interconnections between different groups of biota, acting on varying trophic levels and spatial scales, by grouping the abovementioned soil biota into 'functional groups' (discussed in the following section) in the soil system for nutrient cycling, see Figure 2-3. As shown, soil biota are interdependent which is crucial for both survival as well as delivery of ecosystem processes and functions essential to human well-being.

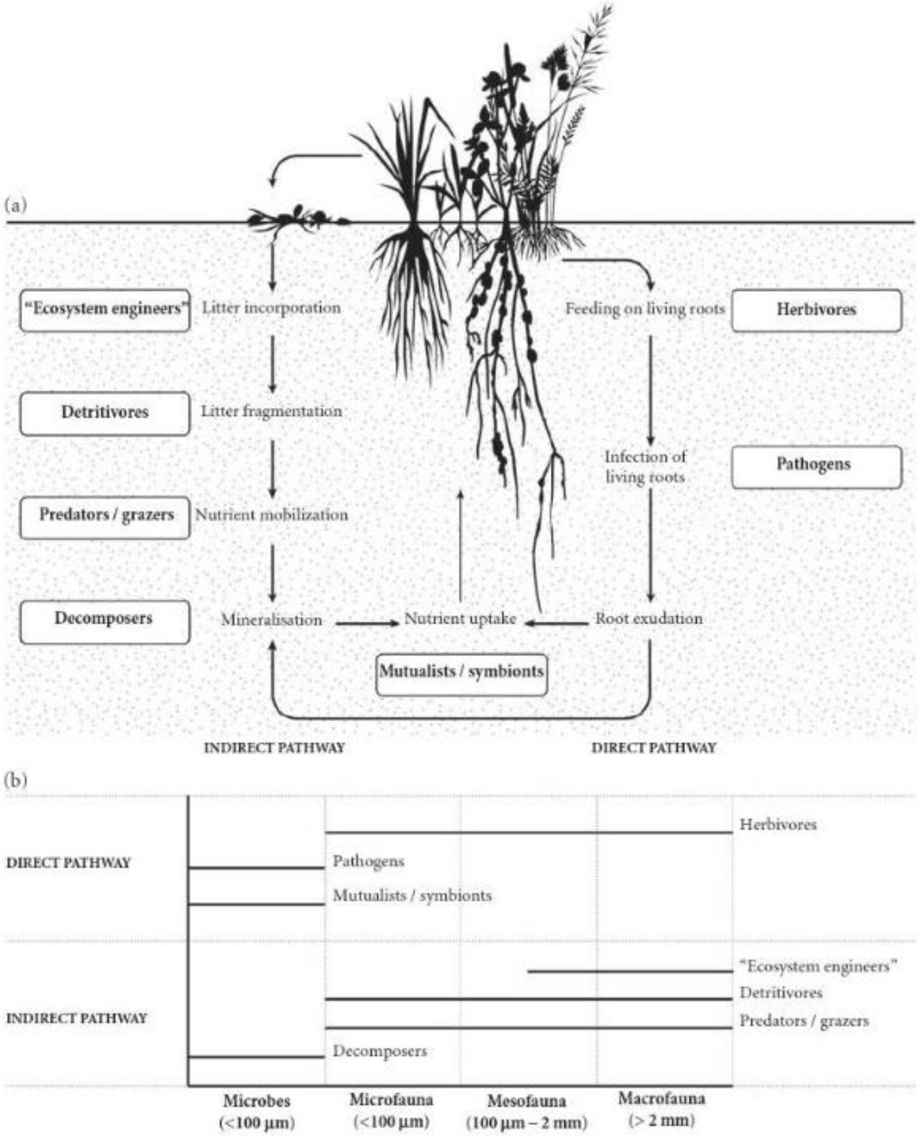


Figure 2-3. a) Soil biota belong to different functional groups (i.e. groups of species with similar traits and effects on processes) involved in carbon and nutrient mobilisation from litter, i.e. dead plant residues ('indirect pathway') and from living plant roots ('direct pathway'). Soil biota have complementary functions and their interactions often increase process rates. b) Some functional groups are restricted to one size class (e.g. microbes), while other functional groups such as detrivores, herbivores, and predators/grazers involve organisms of the micro-, meso- and macrofauna, from (Wurst et al., 2013).

2.2 Soil functions and ecosystem services linkages

According to Faber et al. (2013): '*A major challenge is to link soil biodiversity with soil functioning to assess how soil biodiversity contributes to the delivery of ecosystem services.*' An effort to draw conceptual linkages between soil biodiversity, soil functioning and ecosystem services considered in the soil system is shown in Figure 2-4.

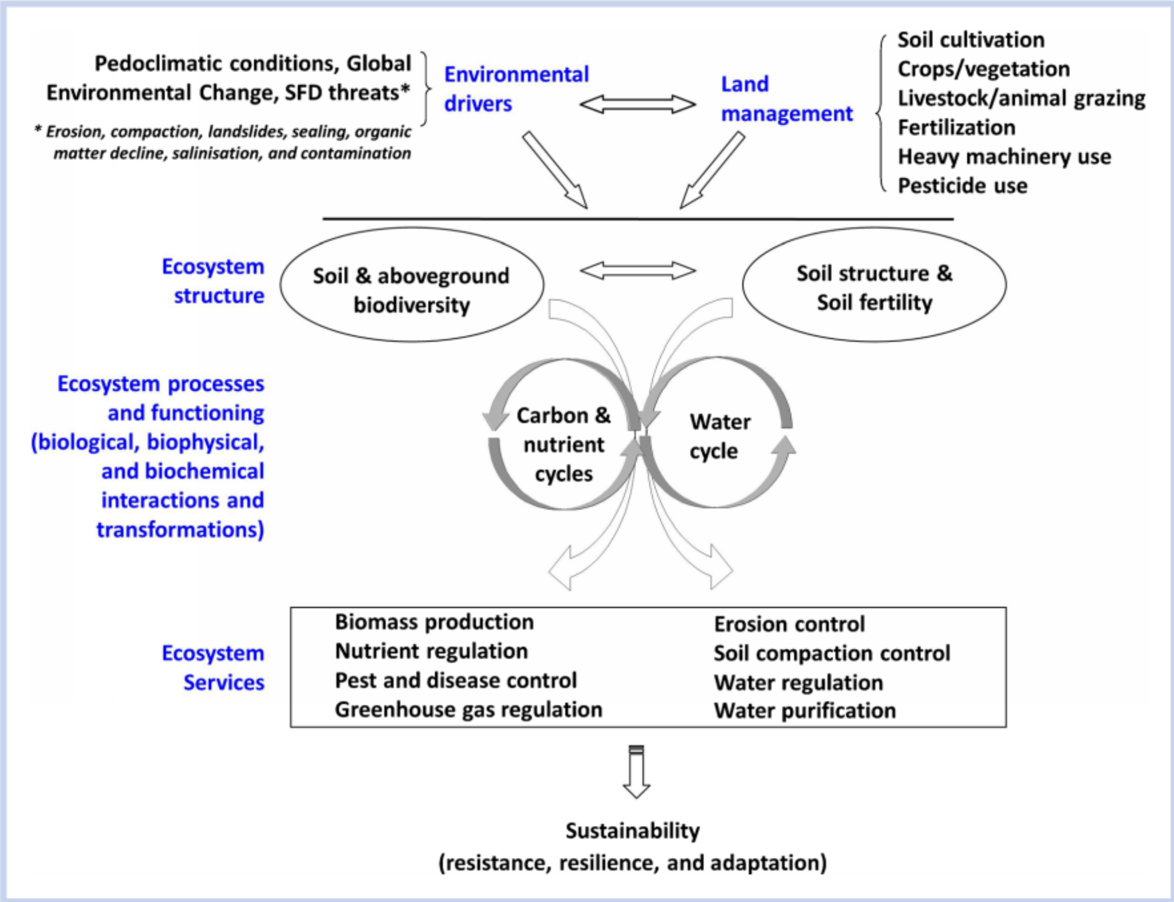


Figure 2-4. Relationships between soil biodiversity, ecosystem functioning and ecosystem services determining ecosystem sustainability and ultimately human well-being, from (Faber et al., 2013) *(adapted from Brussaard et al. 2007). SFD = Soil Framework Directive as proposed in* (EC, 2006).

The delivery of ecosystem services is reliant on a healthy, living soil ecosystem. A mechanistic understanding of the relationships between soil biodiversity and function remains elusive (Ritz et al., 2009; Turbé et al., 2010; Wurst et al., 2013); however, insights into the relationships and linkages between abiotic components, soil biota, and soil processes and functions have been gleaned from comprehensive studies into soil systems. **Ecosystem services** for human benefit are ultimately functional outputs of biological processes resulting from highly complex interactions between the soil biota and the abiotic physical and chemical environment of the soil (Kibblewhite et al., 2008). In aggregate, these **soil** or **ecosystem functions** are provided by assemblages of interacting organisms (i.e. specific groups of the soil biota) (Brussaard, 2013; Kibblewhite et al., 2008), see Figure 2-3 and Figure 2-5. Because of their perceived associations with certain ecosystem functions, groups of related biota interacting with each other and carrying out biological processes, which contribute to these aggregate functions, are often combined into so-called 'functional groups' or 'functional assemblages' (Brussaard, 2013; Kibblewhite et al., 2008). That is, instead of being grouped by size, as such categorisation cannot be unequivocally connected with function (Brussaard, 2013). An important note is that these assemblages do not operate in isolation, but are part of an interactive and interdependent soil system (Kibblewhite et al., 2008; Pulleman et al., 2012; Wurst et al., 2013), as can be seen clearly in Figure 2-2 and Figure 2-3. Indeed, these relatively broad classifications provide generalisations since multiple functions can be performed by different functional assemblages and overlap in functions occurs across all levels (Pulleman et al., 2012), broadly referred to as 'ecological multifunctionality' (Birgé et al., 2016; FAO et al., 2020; Wall et al., 2004).

The USDA's Natural Resources Conservation Service provides extensive resources to understand soil health and assess soil quality largely targeted towards agricultural applications[3]. They maintain that a healthy soil system performs five essential functions (USDA Natural Resource Conservation Service, 2015):

- **Nutrient cycling** – store and cycle nutrients and carbon
- **Water regulation** – regulate and partition water and solute flow
- **Biodiversity and habitat** – sustain biological diversity, activity and productivity
- **Filtering and buffering** – filter, buffer, degrade, detoxify organic and inorganic materials
- **Physical stability and support** – physical stability and support for plant growth (and human structures)

Kibblewhite et al. (2008) synthesised the complex relationships between organisms by establishing four functional assemblages made up of 'key functional groups': 1) decomposers, 2) nutrient transformers, 3) ecosystem engineers, and 4) biocontrollers. Via their associated biological processes and functional attributes, the functional assemblages directly contribute to four key aggregate ecosystem functions that can be linked directly to ecosystem services, see Figure 2-5. They propose that overall soil health (quality) is *a direct expression of the condition of these assemblages, which in turn, depends on the physical and chemical condition of the soil habitat* (Kibblewhite et al., 2008).

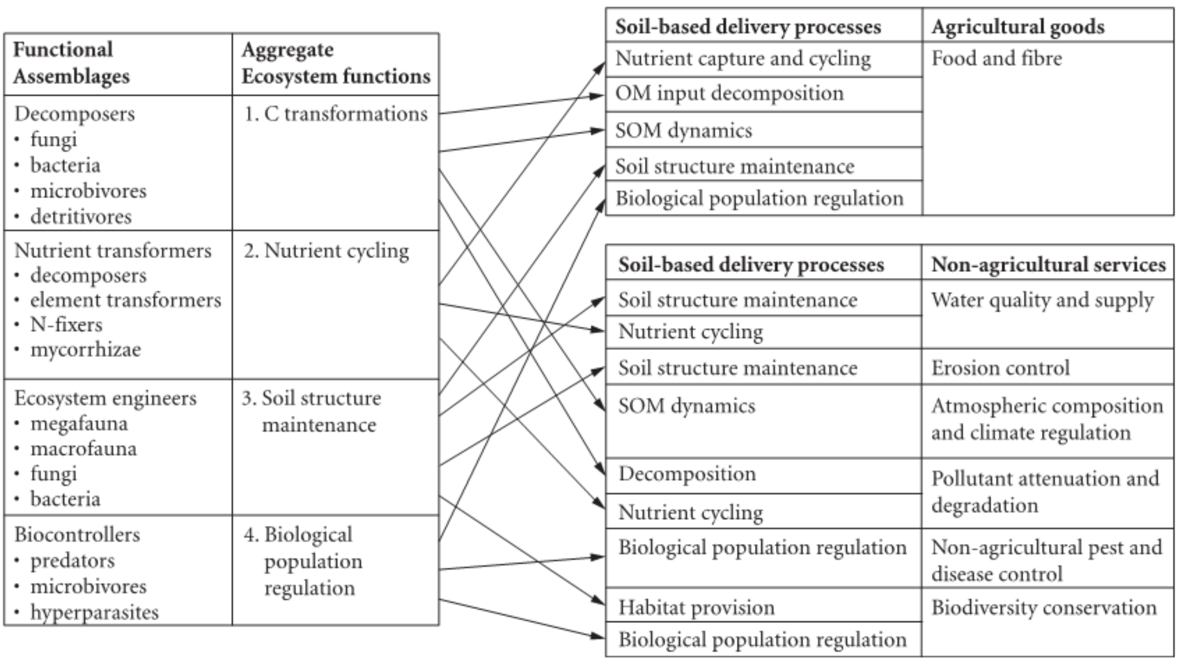


Figure 2-5. Conceptual framework of linkages between soil biota, biologically-mediated soil processes and the provision of soil-based ecosystem goods and services, from (Barrios et al., 2012) *(adapted from* (Kibblewhite et al., 2008)*)*

[3] Soil Health | NRCS Soils (usda.gov)

Shown in Figure 2-5, Kibblewhite et al. (2008) argue that soil health (quality) is fully dependent upon the maintenance of four key functions (i.e. bundles of processes aggregated into ecosystem functions):

- **Carbon transformations –** Transformation of carbon through the decomposition of plant residues and other organic matter, including soil organic matter, together with the synthetic activities of the soil biota, including, and particularly, soil organic matter synthesis. Decomposition in itself is not only an essential ecosystem function and driver of nutrient cycles (i.e. a master variable or 'common currency' that governs microbial activity, and ultimately all soil organisms are driven by energy derived from reduced forms of carbon (Brussaard, 2013)) but also supports a detoxification and waste disposal service. Soil organic matter contributes to nutrient cycling and soil structure maintenance. Sequestration of C in soil also plays some role in regulating the emission of greenhouse gases such as methane and carbon dioxide.

- **Nutrient cycling –** For example, nitrogen, phosphorous and sulphur, including regulation of nitrous oxide emissions. While closely linked to decomposition, the cycling of nutrients is largely mediated by soil microorganisms whose activity levels are regulated by food web interactions within the soil community (Barrios et al., 2012).

- **Soil structure maintenance –** Maintenance of the structure and fabric of the soil by aggregation and particle transport, and formation of biostructures and pore networks across many spatial scales by the combined action of plant roots and soil organisms commonly known as 'soil ecosystem engineers'. This function underpins the maintenance of the soil habitat and regulation of the soil-water cycle and sustains a favourable rooting medium for plants.

- **Biological population regulation –** Biological regulation of soil populations by competition, predation and parasitism, including organisms recognized as pests and diseases of agriculturally important plants and animals as well as humans.

These aggregated ecosystem functions participate in more than one soil-based delivery process, and one or more soil-based delivery processes are required in-turn for the provision of ecosystem goods and services in agricultural landscapes (Barrios et al., 2012).

Classifications of soil organisms can be based on different criteria, and various levels of aggregation have been used between functional approaches (e.g. (Barrios, 2007; Kibblewhite et al., 2008; Wurst et al., 2013)). Addressing this issue, Turbé et al.(2010) provide one of the most extensive reviews of the state of knowledge of soil biodiversity and its contribution to ecosystem services and relevance to human society. They divided the soil organisms according to three 'all-encompassing ecosystem functions': 1) transformation and decomposition (i.e. a combination of carbon transformations and nutrient cycling), 2) biological regulation and 3) soil engineering (i.e. soil structure maintenance) (Turbé et al., 2010). Each of these functions can be performed by assemblages of soil organisms separated into just three broad functional groups (overlapping with those mentioned previously in (Barrios, 2007; Kibblewhite et al., 2008; Wurst et al., 2013)):

1. **Ecosystem engineers** – earthworms, ants, termites and some small mammals modify or create habitats for smaller soil organisms by building resistant soil aggregates and pores. In this way, they also regulate the availability of resources for other soil organisms since soil structures become hotspots of microbial activities.

2. **Chemical engineers** – includes microorganisms (the most abundant soil species) such as bacteria, fungi and protozoans that are responsible for carbon transformation through the decomposition of plant residues and other organic matter as well as transformation of nutrients (e.g. nitrogen, phosphorous, sulphur) made readily available for plants, animals and humans.

   Note: this group is a combination of *decomposers* and *nutrient transformers*.

3. **Biological regulators** – comprises a large variety of small invertebrates, such as nematodes, pot worms, springtails, and mites, which act as predators of plants, other invertebrates, or microorganisms by regulating their dynamics in space and time.

Birgé et al. (2016) created a detailed conceptualisation of the linkages between aboveground-belowground functioning in the rhizosphere that is highly useful for tying these pieces together (Figure 2-6).

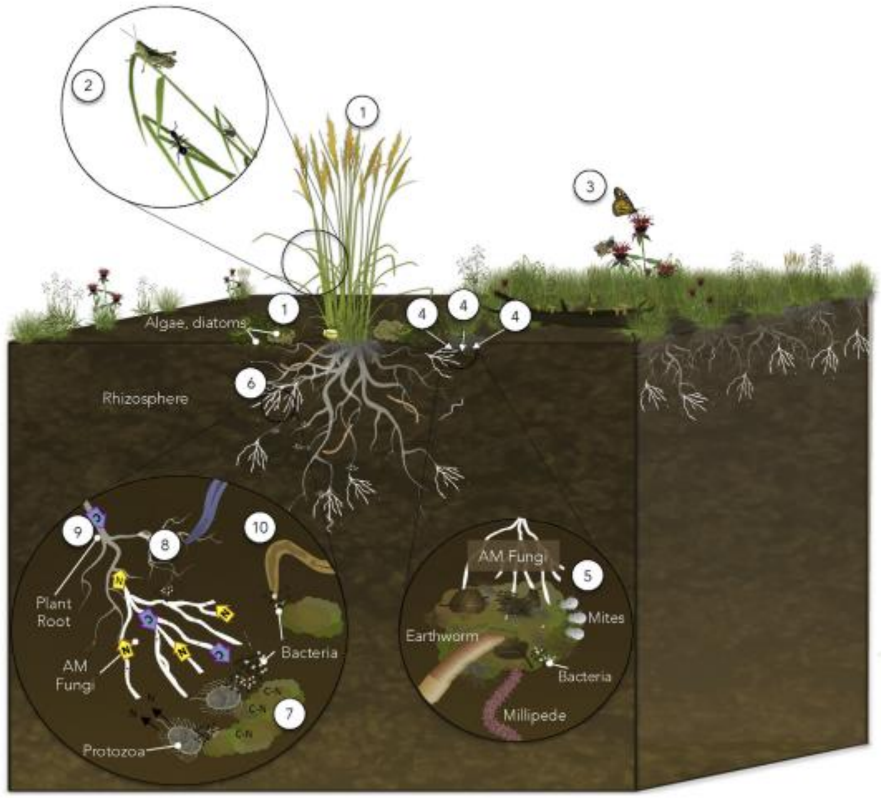


Figure 2-6. A conceptualisation of the tightly coupled aboveground-belowground biodiversity and functioning. Primary productivity (1) is the ultimate source of energy in all ecosystems. Plant materials provide food for a variety of aboveground chewing, sucking, mining (2), and pollinating (3) insects. These plant-insect interactions affect plant chemistry, plant community structure, plant and insect dispersal, and an abundance and diversity of other herbivores and higher trophic levels in the ecosystem (not all shown). Changes in the quantity and/or quality of litter inputs to the soil (4) can result from aboveground herbivory and alter the food source for a variety of belowground detritivores (5). Bacteria, protozoa and arbuscular mycorrhizal (AM) fungi in the rhizosphere (6) directly influence the mineralisation of organic carbon and nitrogen (C-N) stored in humus (7), affecting available nutrients for plants, who may alter fine root turnover (8), and/or release labile carbon (9) to the surrounding microbiota in response, stimulating mineralisation activity, and indirectly influencing higher trophic levels, such as nematodes that feed on roots and bacteria (10). Soil nutrient availability in turn influence plant

community structure, affecting the quality and quantity of litter inputs back to the soil and thus tightening aboveground-belowground diversity and functional linkages. From (Birgé et al., 2016).

There are some differences between how these functional groups are categorised and for which soil functions they are responsible (a challenge inherent to most literature about the soil sciences; even resulting in at least three versions of the diagram shown in Figure 2-5, two of which occurring in different chapters of the same book (Barrios et al., 2012; Brussaard, 2013)), but they seem to converge into a few key groups to which delivery of ecosystem services can be predominantly attributed. There are also many compilations of different 'alternative' soil functions that can be seen in further detail in Volchko (2014) and the informational website "Soil Quality for Environmental Health"[4]. Brief detail on the key functional groups is provided in the following sections.

2.2.1 Ecosystem engineers

The 'ecosystem engineers' include soil macrofauna such as earthworms, termites, ants and enchytraeids (Brussaard, 2013; Pulleman et al., 2012; Wall et al., 2012). This group's keystone species, the earthworms, hold a position of greater importance in the soil ecosystem due to their outsized effect on soil functioning and the provisioning of ecosystem services, see (Blouin et al., 2013; Lavelle, 2013; Lavelle et al., 2006; Turbé et al., 2010; Wall et al., 2012). Due to their biological processes (see previously mentioned reviews for more detail on these mechanisms) they have been directly linked to several invaluable soil functions that enable all others through e.g. pedogenesis, development of soil structure, water regulation, nutrient cycling, primary production, climate regulation, pollution remediation and cultural services (Blouin et al., 2013; Lavelle, 2013; Lavelle et al., 2006). Considering the aggregate soil functions shown in Figure 2-5, Brussaard (2013) states that a major driving force underlying these is soil porosity and associated water-holding capacity, which are in turn important determinants of plant growth, for which organic matter is allocated to feed ecosystem engineers, creating biogenic structures in soil. These functions can be greatly diminished from disturbances and it has been observed that the elimination of earthworm populations due to soil contamination can reduce the water infiltration rate significantly, in some cases even by *up to 93%* (Turbé et al., 2010).

According to Kibblewhite et al. (2008), the key concept of soil health (quality) is that soil provides a living space for the biota, which is defined by the architecture (i.e. space and connectivity) and water content of the pore networks, and gradually decreases from the upper soil strata. Indeed, it is the porous nature of soils that governs so much of their functionality since the physical framework defines the spatial and temporal dynamics of gases, liquids, solutes, particulates and organisms within the matrix, and without such dynamics there would be no function (Kibblewhite et al., 2008). Lavelle (2013) stresses the importance of earthworms and other ecosystem engineers since their ability to dig and burrow and constantly mix the soil is essential to the expansion and maintenance of the structural pore space and to the selection and redistribution of microorganisms. Plant roots also do considerable 'biological drilling', but earthworms are the most efficient bioturbators of the soils, able ale to ingest and egest as solid but complex, organic matter-rich macroaggregates up to 1200 Mg dry soil per hectare and drilling up to 900 m of galleries per m^2. Termites and ants are other powerful ecosystem engineers that complete or replace earthworm activities in most ecosystems where they are present (Lavelle, 2013).

In fact, as stated in Lavelle (2013), the soil habitat imposes a number of constraints on organisms (e.g. available pore space, food source availability) that requires organisms to develop adaptive strategies on in individual basis or via mutualistic associations to survive.

[4] Soil Quality: Soil Functions

These interactions between organisms to adapt to the constraints imposed by the soil environment can be described as 'self-organisation'. Wherein, the energy mobilised through microbial activities and photosynthesis (for plants and their roots) is used by soil ecosystem engineers to build habitats in the compact soil matrix and live there in mutuality with the organisms associated with them (see (Lavelle, 2013) for further elaboration). In agricultural soils, field trials have shown that inoculation of agricultural soils with assemblages of earthworms of different functional types can simultaneously influence a wider array of soil processes than single species inoculation (Barrios, 2007). According to Barrios (2007), this is because the cumulative impacts of vertical and horizontal burrows, surface casting, residue incorporation, and acceleration of plant residue decomposition can lead to improved land productivity even in intensive agriculture production systems.

Box 1. Earthworms (adapted from Pulleman et al. 2012). (Image: R.G. de Goede in Pulleman et al. 2012)

These invertebrates belong to the functional group of ecosystem engineers [3,4]. By producing soil structures such as burrows and excrements they strongly modify the habitat for other soil organisms, including plant roots. Earthworms can play a particularly large role in litter transformation and incorporation as well as soil structure formation [22]. Earthworms are used as bioindicators in contaminated soils because of their sensitivity to soil pollutants (e.g. heavy metals and organic contaminants) [28]. They also respond strongly to agricultural practices (e.g. tillage, crop rotations, pesticides application, organic matter inputs) [22,28,32,37,44,53]. Species (e.g. approximately 100 in France) are classified into three ecological groups (anecics, endogeics and epigeics) that provide different functions and show different sensitivity to soil disturbances or chemical contamination [28,53,32,54]. Epigeic earthworms live at the soil surface and feed on plant litter. Anecics create permanent vertical or subvertical burrows and feed at the soil surface. Those two groups are negatively affected by soil tillage. Endogeics feed on mineral soil enriched in soil organic matter, and therefore benefit from organic matter incorporation either through tillage or the activities of epigeics or anecic earthworms [55]. Anecic and endogeic earthworms play a key role in the formation and maintenance of soil structure, enhance water infiltration and remediation of soil pollutants and reduce soil erosion [30,37]. Total abundance or biomass of earthworms are commonly used as indicators. Nevertheless, the functional group diversity may be a better proxy for habitat quality and soil functions [11,28,53]. An important advantage of earthworms as indicators is that taxonomic identification is relatively easy. Earthworms can be observed with the naked eye and are commonly known and are therefore suitable for communication purposes with stakeholders. However, their spatial variability in the field can be high, which makes representative sampling a laborious task.

Note: Reference numbers correspond to those used in Pulleman et al. 2012, see article for relevant references.

2.2.2 Chemical engineers

This group is a combination of the *decomposers* and *nutrient transformers* functional groups used in Figure 2-5. Microorganisms, especially bacteria and fungi, constitute this group and form the majority of the soil biomass and biodiversity in the soil system and are in fact responsible for providing many of the soil ecosystem services on which human society relies (Gómez-Sagasti et al., 2012). According to Schröder et al. (2018), the most prominent impact of microorganisms on soil quality is their effect on nutrient cycles by fixing or mineralising nutrients from the gross soil nutrient pool thus making them more readily phytoavailable. Well-known mechanisms by which microbes promote nutrient availability include: 1) *biological nitrogen fixation*, by symbiotic N_2-fixing bacteria (e.g. within legume root nodules) and free-living heterotrophic bacteria; 2) *nitrogen mineralisation by fungi*, especially by mycorrhizal fungi which can convert soil organic N to plant-available ammonium; 3) *phosphorous solubilisation*, whereby insoluble organic and inorganic phosphates are transformed into plant-available forms; and 4) *iron solubilisation*, whereby inaccessible ferric ions can be mobilised through microbial activity thus improving iron bioavailability to plants (Schröder et al., 2018). Chemical engineers also play a key role in natural decontamination processes or bioremediation, which has even been regarded as an essential 'regulating ecosystem service' (FAO et al., 2020; Orgiazzi et al., 2016; Turbé et al., 2010).

Plant-associated microbes (e.g. endophytic bacteria, mycorrhiza) can provide a host of benefits directly to plants in a number of ways, including: directly triggering plant health and growth through biosynthesis of various signalling molecules like *phytohormones* (e.g. auxins, cytokinins, ACC-deaminase), biological control of pathogens and modulation of the host plant immune system and resistance to drought and osmotic stress and tolerance to freeze-thaw cycles (Glick, 2012; Schröder et al., 2018). Plant-associated microbes can also directly influence soil structure. One prominent example is arbuscular mycorrhizal fungi which can improve soil aggregation through the production of mycelium, enmeshing and physically protection soil particles, and glomalin, promoting the binding of soil particles (Schröder et al., 2018).

Recent research has shown how mycorrhizal fungi is especially key to the decomposition of organic matter, carbon cycling and ultimately long-term carbon sequestration (e.g. (Clemmensen et al., 2015; Finlay and Clemmensen, 2017; Verbruggen et al., 2016)). To better describe and understand the microbial mechanisms responsible for carbon sequestration, Liang et al. (Jastrow 2017) have proposed a 'soil microbial carbon pump' which is a function of microbial anabolism (i.e. the synthesis of complex molecules in living organisms from simpler ones together with the storage of energy – 'constructive metabolism'). The authors maintain that the microbes enable an 'entombment effect' by which stable soil organic matter (SOM) can be sequestered for long time periods as they build biomass and die-off becoming stabilised 'necromass', see Figure 2-7.

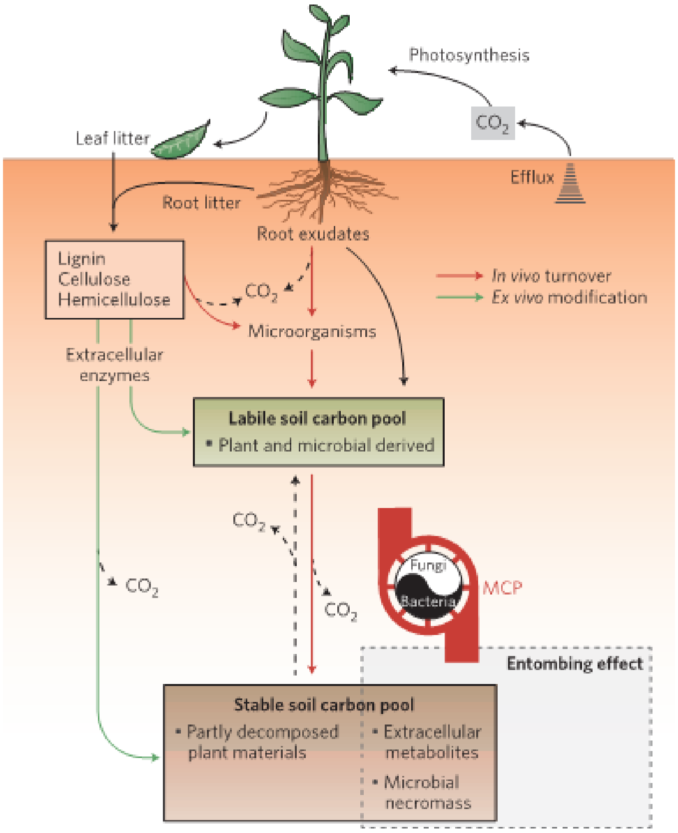
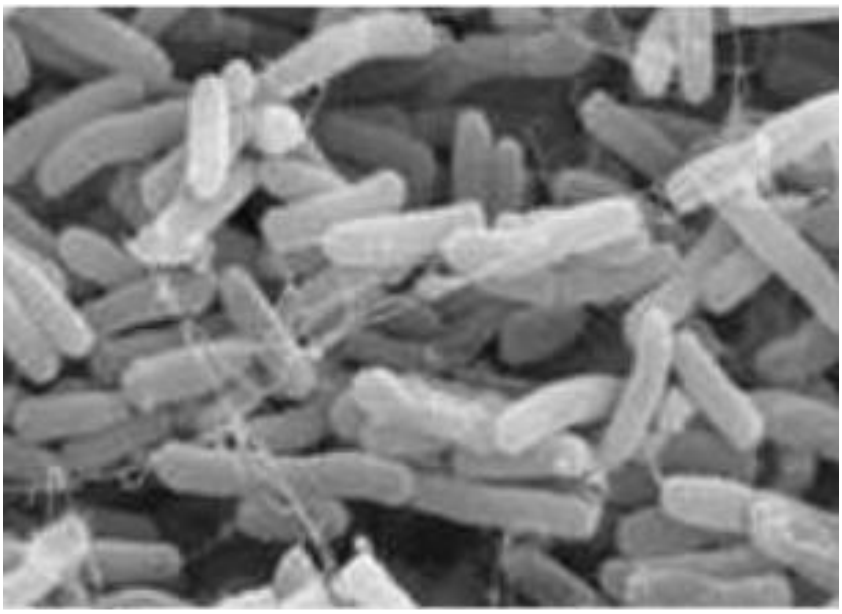
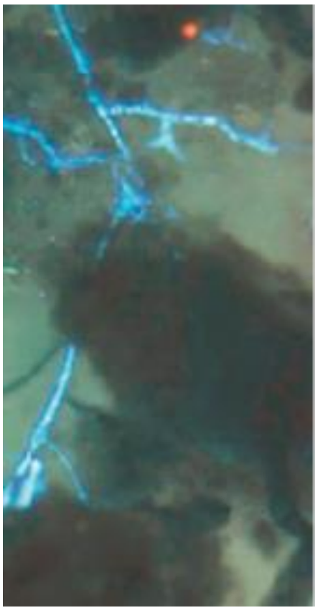


Figure 2-7. Schematic diagram of microbial metabolic processes involved in C cycling in terrestrial ecosystems. Primary production inputs to soils occur through two pathways – in-vivo turnover and ex-vivo modification – that jointly explain soil C dynamics driven by microbial catabolism and/or anabolism before entering the stable soil C pool. Even though the relative importance of in-vivo turnover (red lines) and ex-vivo modification (green lines) varies with different environmental scenarios, the authors argue that the majority of C that is persistent in soils occurs through coupling of the soil microbial carbon pump (MCP – associated with the in-vivo turnover pathway) to stabilisation via the entombing effect. The soil MCP is a conceptual object to demonstrate the fact that microbial necromass and metabolites can be the precursors for persistent soil C, which particularly highlights the importance of microbial anabolism in soil C storage. The yin-yang symbol is used to create a sense of movement and illustrate that the movement is driven, but driven differently, by both bacteria and fungi with different trophic lifestyles. From (Liang et al., 2017).

Regarding contaminated soils, bacteria are typically shown to be more sensitive than fungi to stressors like contamination, which is possibly due to exudates and tolerance strategies to store contaminants like heavy metals in vacuoles (i.e. excluding contaminants from sensitive parts of the cell) (Turbé et al., 2010). According to conventional ecological theory, populations of microorganisms will also be affected by 1) temporal scales – microorganisms have potential for rapid growth and short generation times relative to that of plants and animals; 2) variable activity in microorganisms – active versus inactive or dormant populations, which are those that do not contribute directly to ecosystem processes but are important for resilience of a community to disturbance and might become important as conditions change; 3) competitive strategies like CSR (competitor – stress tolerator – ruderal) strategies – competitors are adapted for rapid resource utilization and long-term site occupation, stress tolerators are adapted to persist in low-resource environments and ruderals are adapted to highly disturbed sites by growing and reproducing quickly; 4) behaviour – changes in cellular processes that occur in response to external signals can be considered 'behaviour', including those triggered by environmental stimuli (e.g. chemotaxis – sensing of, and movement towards, a higher concentration of a required resource) (Prosser et al., 2007).

Box 2. Microorganisms (adapted from Pulleman et al. 2012). Picture on the left side represents bacterial cells, and picture on the right shows (blue-stained) fungal hyphae in soil (Images: K. Ritz in Pulleman et al. 2012)





Chemical engineers decompose organic matter and transform nutrients. Soil microorganisms dominate this functional group [3,4]. They indicate environmental changes by modifications in (i) quantity/biomass, (ii) structure and/or (iii) activity [36,38]. Until now the impact of microbial biomass versus community structure on ecosystem processes and function is uncertain [38,59,60,62]. Levels of functional redundancy among microorganisms depend largely on function and environment considered [15,16,61]. Disconnections between factors driving microbial community structure and those driving its function further complicate indicator selection [62]. To comprehensively assess soil microbial diversity, it is recommended to include indicators of each parameter group: quantity, structure and activity [11]. However, the number of studies and monitoring networks using indicators of all three groups is limited. Different methods [41] are used to describe and quantify microbial diversity at the genotype, phenotype or metabolic level, and thousands of microbial species can occur in just a few grams of soil. To achieve progress in the area of microbial indicators it is important to work on the definition and identification of microbial functional groups and their response to environmental changes [61]. Beside molecular approaches new conceptual models and experimentation are needed to link microbial diversity to ecosystem functions. The development of concepts describing the relationship between the stoichiometry of soil microorganisms (e.g. the C, N and P status) and nutrient cycling is promising [39].

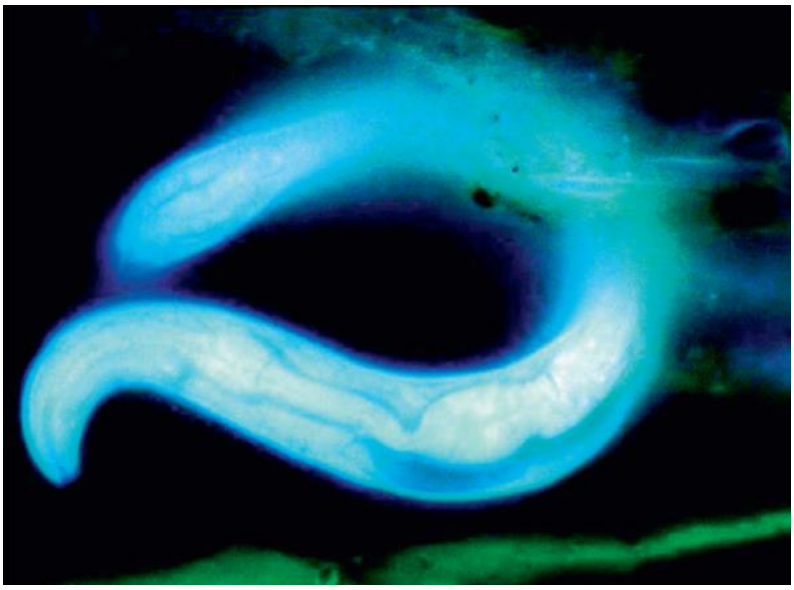
Note: Reference numbers correspond to those used in Pulleman et al. 2012, see article for relevant references.

2.2.3 Biological regulators

Also referred to as *biocontrollers*, this group includes small invertebrates, such as nematodes, springtails and mites, which act as herbivores, predators, grazers, etc. on other invertebrates or microorganisms (Brussaard, 2013; Pulleman et al., 2012; Wall et al., 2012). Soil microfauna (e.g. nematodes) are a diverse group of organisms that mostly feed on bacteria, fungi and (dead) plant materials, and in doing so they can regulate the population size and activity of soil microbes and can promote the competitive ability and dispersal of beneficial rhizosphere microbiota by selective grazing on detrimental soil microorganisms (Wurst et al., 2013). Soil mesofauna (e.g. collembola, mites) also aid in biological regulation by selectively feeding on pathogenic microorganisms, and can aid in distributing smaller soil biota throughout the soil (Wurst et al., 2013). These groups can also contribute to nutrient cycling through feeding on various sources and releasing nutrients via their excrements (Wurst et al., 2013). As stated by

Barrios et al. (2012), the control of soil-borne pests and diseases through biological regulation is an ecosystem service of great economic, human health and environmental importance because global annual crop losses are near 30% and commonly controlled with application of biocides toxic for humans and the environment.

Box 3. Nematodes (adapted from Pulleman et al. 2012). Picture represents a nematode curling through the soil pore space (Image: K. Ritz in Pulleman et al. 2012)



Nematodes are biological regulators and represent one of the most numerous and speciose groups in soils. Soil nematodes are trophically diverse and include economically important plant parasites. They show a high and diverse sensitivity to pollutants and because of their trophic diversity nematode assemblages do not only reflect their own fate, but also the condition of the bacterial, fungal and protozoan communities. These characteristics make them potentially interesting bio-indicators for soil health and soil disturbances [56]. Although nematodes can easily be sampled and extracted from soil, their identification is time consuming and requires expert knowledge. Previous studies demonstrate that the small subunit ribosomal DNA (SSU rDNA) gene harbours enough phylogenetic signal to distinguish between nematode families, genera and often species [57]. A robust and affordable quantitative PCR-based nematode detection tool for agricultural and scientific purposes, and comparable tools for the assessment of the ecological condition of soils, are being developed [58]. Briefly this works as follows: after nematodes extraction from soil the nematode community is lysed and after DNA purification the lysate is used to quantitatively characterize nematode assemblages. The difference in DNA contents of various life stages is limited and different distributions of the life stages barely interfere with quantitative community analyses. Verification in recent field studies suggests that Q-PCR based analysis of nematode assemblages is a reliable alternative for microscopic analysis. The availability of an affordable and user-friendly tool might facilitate and stimulate the use of this ecological informative group of soil inhabitants.

Note: Reference numbers correspond to those used in Pulleman et al. 2012, see article for relevant references.

2.2.4 Vegetation

Amongst the many biological processes associated with soils, Brussaard (2013) states that two are of particular importance: *photosynthesis* (i.e. composition/C fixation, largely occurring aboveground, associated with plant growth) and *respiration* (i.e. decomposition/ C dissipation, largely occurring belowground, inasmuch as associated with plant death). He argues that recognising C as the common denominator and main factor that integrates ecosystem functions implies that the concept of soil functional groups responsible for ecosystem processes that result in ecosystem services cannot be discussed without accounting for a link to the vegetation (Brussaard, 2013). This view stresses the importance of *primary productivity* (the rate of energy capture and carbon fixation by primary producers) as a driver of ecosystem processes and a key determinant of soil biodiversity (Brussaard, 2013; Prosser et al., 2007; Turbé et al., 2010; Wall et al., 2012).

According to Turbé et al. (2010), both the abundance and the quality (i.e. nutritional quality) of vegetation are intricately linked to the diversity of functions performed by soil fauna and flora, since the functional groups contribute to the availability of nutrients and to the soil structure, two crucial parameters for plant growth. Plant biomass production also contributes to regulating the water cycle and local climate through evapo-transpiration (Turbé et al., 2010). The vegetation quality and distribution in the soil matrix is regulated by soil characteristics and soil biodiversity, which ensure the appropriate functioning of the ecosystem, providing the conditions for plant growth (Turbé et al., 2010). The inverse is also true, and there have been demonstrable positive effects by vegetation on the soil as a habitat for organisms; for example, in the case of agroforestry applications (Barrios et al., 2012), crop selection in agricultural settings (Cavigelli et al., 2013; Wall et al., 2012) and short-term rotation coppicing (SRC) with diverse clones of willows and poplar (Baum et al., 2009; Müller et al., 2018).

Another important factor demonstrating the influence of vegetation on soil biota is the issue of 'hotspots' of biological activity, which Barrios (2007) notes arise due to the distribution of soil biota in space and time not being random or homogeneous but rather in concentrated pockets of activity that are mostly associated with the availability of C substrates. These 'hotspots' include the rhizosphere and AMF hyphosphere, biogenic structures (i.e. soil aggregates), soil C pools (i.e. light fraction SOM), and organic detritus (i.e. litter), where key functional assemblages can be studied to focus on soil biological processes that underpin the provision of soil-based ecosystem services (Barrios, 2007; Barrios et al., 2012). The general consensus is that soil biological processes are not randomly distributed but are largely aggregated near C substrates, and that greater knowledge about plant–soil biota interactions have great potential to improve understanding of the impacts of soil biota at larger scales (Barrios, 2007; Barrios et al., 2012).

To better understand the role of vegetation in the ecosystem, Thijs et al. (2016, 2017) describe a model of a 'metaorganism' (host and microbiome together) through which to gain a greater understanding of the synergistic actions of plants and microorganisms and plant-microbial functions (in the case of phytoremediation). Similarly, Liu et al. (2019) envision the 'eco-holobiont' – the microbiome of an entire ecosystem, including feedbacks from microbiomes associated with biotic (e.g., plants and animals) and abiotic (e.g., soil) components and their inter-action, which are critical for shaping host-associated micro-biomes and their role in host fitness and ecosystem health. They further argue that crop diversity and rotation should be encouraged as it may have positive influences on soil microbiome and pollinator health (Liu et al., 2019)

2.2.5 The role of diversity

The functional group approach, rather than an exhaustive accounting of all manner of soil organisms in the system, is widely considered to be a pragmatic and effective way to study the linkages between potentially manageable soil biota and tangible functions that underpin ‘soil-based’ ecosystem services (Barrios, 2007; Barrios et al., 2012; Kibblewhite et al., 2008); however, the functional group-based approach is not without criticism. Brussaard (2013) discusses the differences between a 'soil biogeochemistry' perspective on ecosystem functioning, which downplays the importance of functional groups, versus a 'soil biology' view, which considers such details as a necessary perspective, and their accompanying ecological perspectives. Brussaard (2013) further mentions three specific issues with functional groupings: 1) some functional groups are, like body-size groups (shown above), associated with more than one aggregate function, 2) different life-stages of the same species may be associated with different functions, and 3) within one group, species may occur with different effects on soil functions (e.g. bacteria and fungi). Additionally, some studies have shown 'functional redundancy' within the various functional groups by removing species from controlled soil communities and tracking the resulting effects on soil functioning (Barrios, 2007; Wurst et al., 2013). Results indicate that certain species have an overriding importance in their functional group (i.e. 'keystone species') while others could be removed and still retain stability for certain essential soil functions (i.e. 'redundant species') (Barrios, 2007). These diversity-function relationship studies indicate that community composition, especially the traits (i.e. well-defined properties of organisms) of key species or groups, their relative abundance and interactions with other groups, appears to be the most significant driver of soils processes and functions, rather than 'species richness' (i.e. biodiversity) per se (Barrios, 2007; Brussaard, 2013; Wurst et al., 2013). Put simply, 'functional diversity' (i.e. number of functional traits) may actually be a more important factor than simply 'species diversity' (i.e. number of taxonomically distinct organisms) (Brussaard, 2013; Wurst et al., 2013). For example, to measure the effectiveness of phytoremediation to remediate and restore soil quality, Epelde et al. (2008) suggest that functional diversity is a key indicator of soil health that can be measured by e.g. community-level physiological profiling (CLPP).

This refined 'trait-based ecology' approach can provide a sound scientific basis for the choice of species or cultivars (i.e. by judiciously selecting plant species or cultivars with 'key characteristics') to maintain both aboveground productivity and belowground ecosystem services in sustainable 'agro-ecosystems' (Brussaard, 2013). Although, robust, supportive models are still lacking.

Schröder et al. (2018) provide an interesting discussion regarding the effects of microbial diversity on functioning (exact wording as in source, emphasis added, (Schröder et al., 2018) pp. 1116):

> "From all the arguments listed above, it becomes clear that soil microbes contribute to a very significant extent to plant growth on marginal soils. On the other hand, soil amendments that favour microbial activity also have the potential to increase plant growth, through increased mineralisation, resistance to plant disease (induced systemic resistance), or drought (induced systemic tolerance) and all other aspects associated with beneficial plant-microbe interaction. As a general rule, we may assume that the more microbes are active, the more they will contribute to soil mineralisation processes. Microbes are however sensitive to environmental conditions such as water content, pH or temperature. Hence microbially controlled soil processes are likely to be unstable in a versatile environment, and the loss of a species may lead to the loss of a given soil function. This is where microbial diversity is of importance: the higher it is, the more likely that the loss of a given species (because of a disturbance) is compensated by another one similar in functionality. **In this case, this is not the taxonomic diversity per se (Estendorfer et al., 2017) that matters, but rather the functional diversity, defined as the range of processes that a microbial community can contribute to** (Heemsbergen, 2004). To measure the contribution of microbial communities in soil processes, both, taxonomic and functional diversity need to be taken into account. High taxonomic diversity could therefore lead to higher stability and resilience of soil processes only if functional redundancy in the community is high. Reversely, some soil processes are dominated by single or a few individual species and therefore the rate of these processes will depend on species identity rather than high functional diversity (Gamfeldt et al., 2008). **Hence, a functional trait (such as mineralization and nitrogen fixation) can be a better ecological indicator of soil microbiological quality than the abundance of specific taxa**" (Schröder et al., 2018).

Functional groups, functional redundancy, functional diversity, traits and relationships between diversity and function are complex issues and still widely debated, see (Barrios, 2007; Brussaard, 2013; FAO et al., 2020; Kibblewhite et al., 2008; Orgiazzi et al., 2016; Wall et al., 2012; Wurst et al., 2013) for further reading. Also, Turbé et al. (2010) provide a clear explanation of the issues involved in functional redundancy, shown below in Figure 2-8.

Box 4: Functional redundancy: myth or reality?

Although there are many reasons to protect biodiversity for its intrinsic value, conservation efforts are increasingly justifying biodiversity conservation for the functions, or services, it provides. In this case, a major question is whether all species are important for soil ecosystem functioning.

To date, no consistent relationship between soil species diversity and soil functions has been found (Bardgett 2002; Bardgett 2005), implying that more species do not necessarily provide more services. This is because several species can perform the same function. Thus, according to the 'redundant species' hypothesis, only a minimum number of species is necessary for soil ecosystems to function (Naeem, Thompson et al. 1995) and the loss of a functionally redundant species would have little impact on the quantity or quality of the service provided (Naeem, Thompson et al. 1995; Hunt and Wall. 2002).

Other theories pertain that the fact that many soil species may appear functionally 'redundant' is rather related to our lack of understanding of soil systems (Wolters 2001). Indeed:

- **Not all functions exhibit redundancy,** some species may be the only ones able to perform their function. For instance, many species are involved in the decomposition of organic matter, and the loss of one of these species may not necessarily have a direct negative effect on the functioning of the ecosystem. In contrast, the breakdown of some toxic chemicals may only be performed by a single species of bacteria, in which case, the loss of this species means a complete loss of the function in the ecosystem. Nitrification (that is the transformation of nitrite into nitrate) is also performed by very few microorganisms.
- **Redundancy is highly context-dependent,** for instance, while two species of bacteria may appear to perform the same decomposition function, they may not perform it under the same range of conditions, or at all times. For example, one species could become inactive under heat stress whereas the other could still be functioning perfectly, or may even show increased activity.
- **Soil organisms can contribute to more than one function, f**or example, many species of fungi and bacteria that are responsible for most of the transformation and decomposition processes also contribute, albeit to a lesser extent, to soil structure modification. Moreover, because of the integrated nature of soil food webs, some 'redundant' species may gain functional significance by regulating the activity of a functionally important species. Thus, species that are redundant for one function may play a key functional role elsewhere in the food web.

Therefore, according to the 'insurance hypothesis', it seems that there are many ways in which current, apparently redundant, diversity may have a function under future, unpredictable conditions. Given that we still know little about the role of single species, and according to the precautionary principle, it may thus be important to preserve this biodiversity for insurance purposes and not put our future at stake by reducing the insurance value of the biodiversity capital. This is also consistent with the principle of 'no net loss of biodiversity' (whether in the quantity or quality of the functions provided), advocated by the Convention on Biological Diversity.

Figure 2-8. Functional redundancy: myth or reality, from (Turbé et al., 2010)*, pp. 42.*

2.3 Soil-based ecosystem services

To briefly reiterate, **soil functions** is used to define the biological, geochemical and physical processes and components that take place within a soil or larger ecosystem, i.e. underlying processes maintaining the ecosystem, and **ecosystem services** encompass the tangible and intangible benefits that humans obtain from ecosystems (Bünemann et al., 2018; Orgiazzi et al., 2016). Many ecosystem services can be intuitively linked to the functioning of the soil biota and their interactions within their physical and chemical environment (Brussaard, 2013; Dominati et al., 2010; Faber and Van Wensem, 2012; Thomsen et al., 2012). Extensive lists of soil services have been by covered by many different authors (e.g. (Brussaard, 2013; Dominati et al., 2010; Haygarth and Ritz, 2009; Robinson et al., 2013; Wall et al., 2004)) with both considerable differences in terminology and overlap between the many variations. A few of these proposals will be briefly discussed in this section.

One of the earliest (before the Millennium Ecosystem Assessment) and more frequently cited sets of soil-based ecosystem services comes from Wall et al. (2004), including 16 ecosystem services provided by soil and sediment biota as selected by a consortium of scientists for the Committee on Soil and Sediment Biodiversity and Ecosystem Functioning (SSBEF):

1. Regulation of major biogeochemical cycles
2. Retention and delivery of nutrients to plants and algae
3. Generation and renewal of soil and sediment structure and soil fertility
4. Bioremediation of wastes and pollutants
5. Provision of clean drinking water
6. Modification of the hydrological cycle
7. Mitigation of floods and droughts
8. Erosion control
9. Translocation of nutrients, particles and gases
10. Regulation of atmospheric trace gases (e.g. CO_2, NOx) (production and consumption)
11. Modification of anthropogenically driven global changes (e.g. carbon sequestration, modifiers of plant and algae responses)
12. Regulation of animal and plant (including algae, macrophytes) populations
13. Control of potential pests and pathogens
14. Contribution to plant production for food, fuel and fibre
15. Contribution to landscape heterogeneity and stability
16. Vital component of habitats important for recreation and natural history

Another frequently cited example is from Haygarth and Ritz (2009) who provide a framework based upon 18 critical ecosystem services to give focus and highlight specific areas that require priority in the short- and longer-term future in the UK, see Table 2-2. They are concerned that anthropogenically induced changes in land use or management will result in soils not being utilised to provide the functions to which they are best suited. For example, soils primarily suited for food supply may be given over to provide a platform for construction. This is an allpervading and recurring concern and highlights the importance of critical decisions, thresholds and potential 'tipping points.' Meaning that once critical soil functions are lost, they are irrecoverable, potentially for millennia, representing a loss of resource that is fundamental to the UK's (and the world at-large) national livelihood and well-being (Haygarth and Ritz, 2009).

Table 2-2. Ecosystem services and functions appropriate to soils and land use in the UK, adapted from (Haygarth and Ritz, 2009).

Category	Ecosystem Service	Soil function	Example
Supporting			
1	Primary production	Support for terrestrial vegetation	Support for principal photoautotrophs
2	Soil formation	Soil formation processes	Weathering of rock and accumulation of organic material
3	Nutrient cycling	Storage, internal cycling and processing of nutrient	N-fixation and N and P mineralisation and cycling
Provisioning			
4	Refugia	Providing habitat for resident and transient populations	Burrows for soil macrofauna
5	Water storage	Retention of water in landscape	Retention of water in pore network, modulates soil biochemical processes
6	Platform	Supporting structures	Supporting housing, industry, infrastructure
7	Food supply	Provisioning plant growth	Provisioning for crops and livestock for farming
8	Biomaterials	Provisioning plant growth	Producing timber, fibre, fuel
9	Raw materials	Provisioning source materials	Topsoil, mineral, aggregates extraction
10	Biodiversity and genetic resources	Sources of unique biological materials and products	Medical products, genes for resistance to pathogens and pests
Regulating			
11	Water quality regulation	Filtration and buffering of water	Potable water for human consumption and good ecological status of rivers, lakes and seas
12	Water supply regulation	Regulation of hydrological flows	Flood control where surplus, irrigation where deficit
13	Gas regulation	Regulation of atmospheric chemical composition	CO_2/O_2 balance, O_2 for UVB protection, and SO_x levels
14	Climate regulation	Regulation of global temperature, precipitation, and other biologically mediated climatic processes	Greenhouse gas regulation
15	Erosion control	Soil and colloid retention within an ecosystem	Retention of soil on hillslopes and in wetlands
Cultural			
16	Recreation	Providing a platform for recreational activities	Eco-tourism, sport
17	Cognitive	Opportunities for non-commercial activities	Aesthetic, education, spiritual, scientific value
18	Heritage	Holds archaeological record of terrestrial occupancy and civilisations	Preservation/destruction of archaeological record

Regarding specifically soil-based (or agroecosystem-based) ecosystem services, Dominati et al. (2010) provide a detailed overview of current thinking on, and approaches for, the classification and quantification of soil natural capital and ecosystem services. They base their work on a scientific understanding of soil formation, functioning and classification systems to develop a framework consisting of five interconnected components: 1) soil natural capital, characterised by standard soil properties well known to soil scientists; 2) the processes behind soil natural capital formation, maintenance and degradation; 3) drivers (anthropogenic and natural) of soil processes; 4) provisioning, regulating and cultural ecosystem services; and 5) human needs fulfilled by soil ecosystem services (Dominati et al., 2010). In this framework, ecosystem services are defined as *the beneficial flows arising from natural capital stocks and fulfilling human needs*. These include provisioning (food, wood and fibre; physical support; raw materials), regulating (flood mitigation; filtering of nutrients; biological control of pests and diseases; recycling of wastes and detoxification; carbon storage and regulation of N2O and CH4 emissions) and cultural services (spirituality; knowledge; sense of place; aesthetics). See Figure 2-9 for a graphic overview of this framework and the ecosystem services included therein.

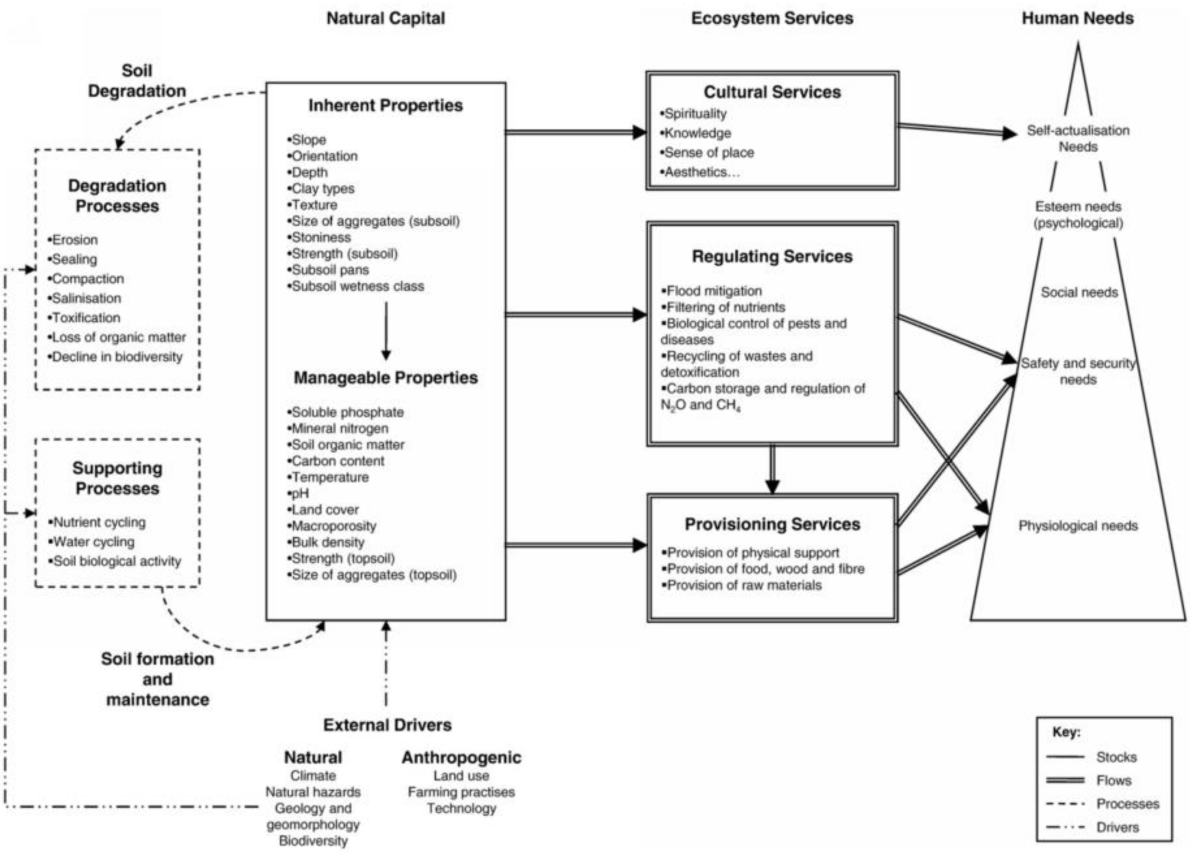


Figure 2-9. Framework for the provision of ecosystem services from soil natural capital, from (Dominati et al., 2010).

Turbé et al. (2010) note that most of the ecosystem services provided by soils are supporting services, or services that are not directly used by humans, but underlie the provisioning of all other services. These include nutrient cycling, soil formation and primary production, as shown in the previous examples. In addition, soil biodiversity influences the main regulatory services, namely the regulation of atmospheric composition and climate, water quantity and quality, pest and disease incidence in agricultural and natural ecosystems, and human diseases. Soil organisms may also control or reduce environmental pollution (e.g. via bioremediation). Soil organisms also contribute to provisioning services that directly benefit humans. For example, the genetic resources of soil microorganisms can be used for developing novel pharmaceuticals. According to Turbé et al. (2010), the contributions of soil biodiversity, in terms of soil-based ecosystem services, can be grouped under the six following aggregated categories:

1. **Soil structure, soil organic matter and fertility** – soil organisms are affected by but also contribute to modifying soil structure and creating new habitats. Soil organic matter is an

important ‘building block’ (or 'common currency' (Kibblewhite et al., 2008)) for soil structure, contributing to soil aeration, and enabling soils to absorb water and retain nutrients. All three functional groups (i.e. ecosystem engineers, chemical engineers and biological regulators) are involved in the formation and decomposition of soil organic matter, and thus contribute to structuring the soil. For example, some species of fungi produce a protein (glomalin) which plays an important role in soil aggregation due to its sticky nature. Also, nutrients are released and rendered bioavailable for plants and other organisms by soil organisms via the decomposition of soil organic matter. The residual soil organic matter forms humus, which serves as the main driver of soil quality and fertility. As a result, soil organisms indirectly support the quality and abundance of plant primary production (Turbé et al., 2010).

2. **Regulation of carbon flux and climate control –** soil organisms process 25,000 kg of organic matter per year per hectare of soil. Soil organisms increase the soil organic carbon pool through the decomposition of dead biomass, while their respiration releases carbon dioxide (CO2) to the atmosphere. Carbon can also be released to the atmosphere as methane, a much more powerful greenhouse gas than CO2, when soils are flooded or clogged with water. The loss of soil biodiversity will reduce the ability of soils to regulate the composition of the atmosphere, as well as the role of soils in counteracting global warming (Turbé et al., 2010).

3. **Regulation of the water cycle –** soil ecosystem engineers affect the infiltration and distribution of water in the soil, by creating soil aggregates and pore spaces. Soil biodiversity may also indirectly affect water infiltration, by influencing the composition and structure of the vegetation, which can protect the soil surface, influence the structure and composition of litter layers and influence soil structure by rooting patterns. The diversity of microorganisms in the soil contributes to water purification, nutrient removal, and to the biodegradation of contaminants and of pathogenic microbes. Plants also play a key role in the cycling of water between soil and atmosphere through their effects on evapo-transpiration. The loss of this service will reduce the quality and quantity of ground and surface waters as nutrients and pollutants (e.g. pesticides) may no longer be degraded or neutralised. Surface runoff will likely also increase, thereby increasing the risks of erosion and even landslides in mountain areas, and of flooding and excessive sedimentation in lowland areas (Turbé et al., 2010).

   Another major factor controlling the water infiltration rate in soil and its capacity for water retention is whether the surface of soil is covered with vegetation or plant litter. The presence of vegetation can regulate the quantity of water reaching the soil by protecting it with leaves, capturing the water and structuring the soil with underground roots. In effect, water is kept locally and can permeate into underground reserves. When vegetation is limited or absent, water will run off, instead of being absorbed, enhancing the erosion of soil particles. Plant roots prevent soil particles from being washed away with water flows, keep soil macro-aggregates together and can prevent landslides (Turbé et al., 2010).

4. **Decontamination and bioremediation** – chemical engineers play a key role in bioremediation, by accumulating pollutants in their bodies, degrading pollutants into smaller, non-toxic molecules, or modifying those pollutants into useful metabolic

molecules. Humans can utilise these remediation capacities of soil organisms to directly engineer bioremediation, whether in situ or ex situ, or by promoting microbial activity (e.g. by adding soil amendments. Phytoremediation, which is indirectly mediated by soil organisms, is also useful to remove persistent pollutants and heavy metals (Turbé et al., 2010).

All the abiotic processes involved in soil decontamination and their effectiveness are determined by the physico-chemical properties of soil surface, soil porosity, the chemical properties of pore-water compartment, and, of course, the physico-chemical properties of the pollutants (e.g. behaviour of organic and inorganic molecules may be significantly different in the soil matrix). The presence of active surface fractions such as organic matter, possessing high surface areas and charges can, for example, facilitate oil retention in the soil matrix. All these physico-chemical properties are directly or indirectly linked to soil properties and biodiversity. For example, earthworms and microbes are key actors in the determination of soil aggregation and porosity. Similarly, microbial activity can locally alter soil pH, affecting soil aggregation and its capacity to absorb contaminants. Therefore, a high diversity and biological activity within soils, especially at the level of chemical engineers, but also in the case of ecosystem engineers, is indispensable to ensure this crucial service through a direct influence on soil biotic degradation processes and an indirect influence on soil abiotic degradation processes of pollutants (Turbé et al., 2010).

5. **Pest control** – soil biodiversity promotes pest control, either by acting directly on belowground pests, or by acting indirectly on aboveground pests. Pest outbreaks occur when microorganisms or regulatory soil fauna are not performing efficient control. Ecosystems presenting a high diversity of soil organisms typically present a higher natural control potential, since they have a higher probability of hosting a natural enemy of the pest. Interestingly, in natural ecosystems, pests are involved in the regulation of biodiversity. Soil-borne pathogens and herbivores control plant abundance, which enhances plant diversity. In natural ecosystems, the loss of pathogenic and root-feeding soil organisms will cause a loss of plant diversity and will enhance the risk of exotic plant invasions. Changes in vegetation also influence aboveground biodiversity. Loss of this ecosystem service, therefore, will cause loss of biodiversity in entire natural ecosystems (Turbé et al., 2010).

6. **Human health** – soil organisms, with their astonishing diversity, are an important source of chemical and genetic resources for the development of new pharmaceuticals. For instance, many antibiotics used today originate from soil organisms. Soil biodiversity can also have indirect impacts on human health. Land-use change, global warming, or other disturbances to soil systems can release soil-borne infectious diseases and increase human exposure to those diseases. Disturbed soil ecosystems may also lead to more polluted soils or less fertile crops, all of which can indirectly affect human health, for example through ingestion of contaminated food (Turbé et al., 2010).

In the definition of these six services, Turbé et al. (2010) grouped several sub-services into one aggregated service. They also provide a useful comparison of their aggregated ecosystem services to those defined in the Millennium Ecosystem Assessment (MEA) report (The Millennium Ecosystem Assessment, 2005), see Table 2-3.

Table 2-3. Comparison of the aggregated ecosystem services proposed by Turbé et al. (2010) *to MEA nomenclature, from* (Turbé et al., 2010).

Aggregated services	MEA nomenclature	Category of service
Soil organic matter recycling and fertility, including soil formation	Decomposition, nutrient cycling, soil formation, *primary production*, erosion regulation	Supporting and Provisioning
Regulation of carbon flux and climate control	Climate regulation	Regulating
Water cycle regulation	Water regulation and water purification	Regulating
Decontamination and bioremediation	-	Regulating
Pest control	Disease regulation	Regulating
Human health	Disease regulation	Regulating

As can be seen, **a major challenge is to find a widely agreed upon selection of soil functions and ecosystem services that can be used as the standard for soil ecosystem services**. The terminology also differs with each article, research group and classification effort. For example, the Landmark project[5], a Europe-wide research project on the sustainable management of land and soil in Europe, refers to soil functions as simply "soil-based ecosystem services", encompassing the following five functions relating to agriculture:

1. **Primary productivity** – the capacity of a soil to produce plant biomass for human use, providing food, feed, fibre and fuel within natural or managed ecosystem boundaries.
2. **Water purification and regulation** – the capacity of a soil to remove harmful compounds from the water that it holds and to receive, store and conduct water for subsequent use and the prevention of both prolonged droughts and flooding and erosion.
3. **Climate regulation and carbon sequestration** – the capacity of a soil to reduce the negative impact of greenhouse gas (i.e. CO_2, CH4, and N2O) emissions on climate.
4. **Soil biodiversity and habitat provisioning** – The multitude of soil organisms and processes, interacting in an ecosystem, making up a significant part of the soil's natural capital, providing society with a wide range of cultural services and unknown services.
5. **Provision and cycling of nutrients** – The capacity of a soil to receive nutrients in the form of by-products, to provide nutrients from intrinsic resources or to support the acquisition of nutrients from air or water, and to effectively carry over these nutrients into harvested crops.

Bünemann et al. (2018) recommend that to better consolidate the ecosystem services in focus throughout these various schemes and frameworks, they can be seen as a soil-related sub-set of the ecosystem services mentioned in the Common International Classification of Ecosystem Services (CICES[6]) (Bünemann et al., 2018). For example, as shown below in Table 2-4, researchers developing the BISQ (Biological Indicators for Soil Quality) soil assessment framework aligned their 11 proposed soil services (selected via weighted multi-criteria analysis in a workshop with experts) with aggregated CICES categories (Rutgers et al., 2014, 2012).

[5] http://landmark2020.eu/soil-functions-concept/
[6] https://biodiversity.europa.eu/maes/common-international-classification-of-ecosystem-services-cices-classification-version-4.3

Table 2-4. Soil ecosystem services in the BISQ soil assessment framework aligned with CICES classification, summarised from (Rutgers et al., 2014) *and updated to match divisions in CICES v 5.1.*

Soil service in support of ecosystem service	Aggregated soil ecosystem service	CICES section	CICES (v 5.1) division
1a. Nutrient retention and release	1. Supporting the production functions of the soil (crop, cattle, landscape) for agriculture, forests, nature, recreation and green areas	Provisioning services	Biomass; Water; Genetic materials
1b. Soil structure			
1c. Natural disease suppressiveness ('biocontrol')			
2a. Resistance and resilience	2. Resistance, resilience and flexibility (general support function)	Regulation and maintenance services (CICES), regulating services (TEEB), regulating and supporting services (MEA)	Transformation of biochemical or physical inputs to ecosystems; Regulation of physical, chemical, or biological conditions
2b. Potential for other land uses			
3a. Fragmentation and mineralisation of plant residues, building of soil organic matter, carbon cycling	3. Supporting the regulation functions of the soil (incl. Nutrient cycles, clean ground water and surface water)		
3b. Natural attenuation or purifying capacity and nutrient cycling			
3c. Water: retention, release and transport			
3d. Climate functions (all temporal and spatial scales)			
4a. Habitat function, biodiversity and gene pool	4. Supporting soil biodiversity and habitat functions	Cultural services	Direct, in-situ and outdoor interactions with living systems that depend (or do not) on presence in the environmental setting
4b. Ethical, cultural and educational functions			

Similarly, one of the clearest delineations of soil-based ecosystem services (derived from the Millennium Ecosystem Assessment) is presented in the Global Soil Biodiversity Atlas (Orgiazzi et al., 2016) by linking them to correlated soil/ecosystem functions and soil biota (Figure 2-10), which was derived from the conceptual framework shown in Figure 2-5.

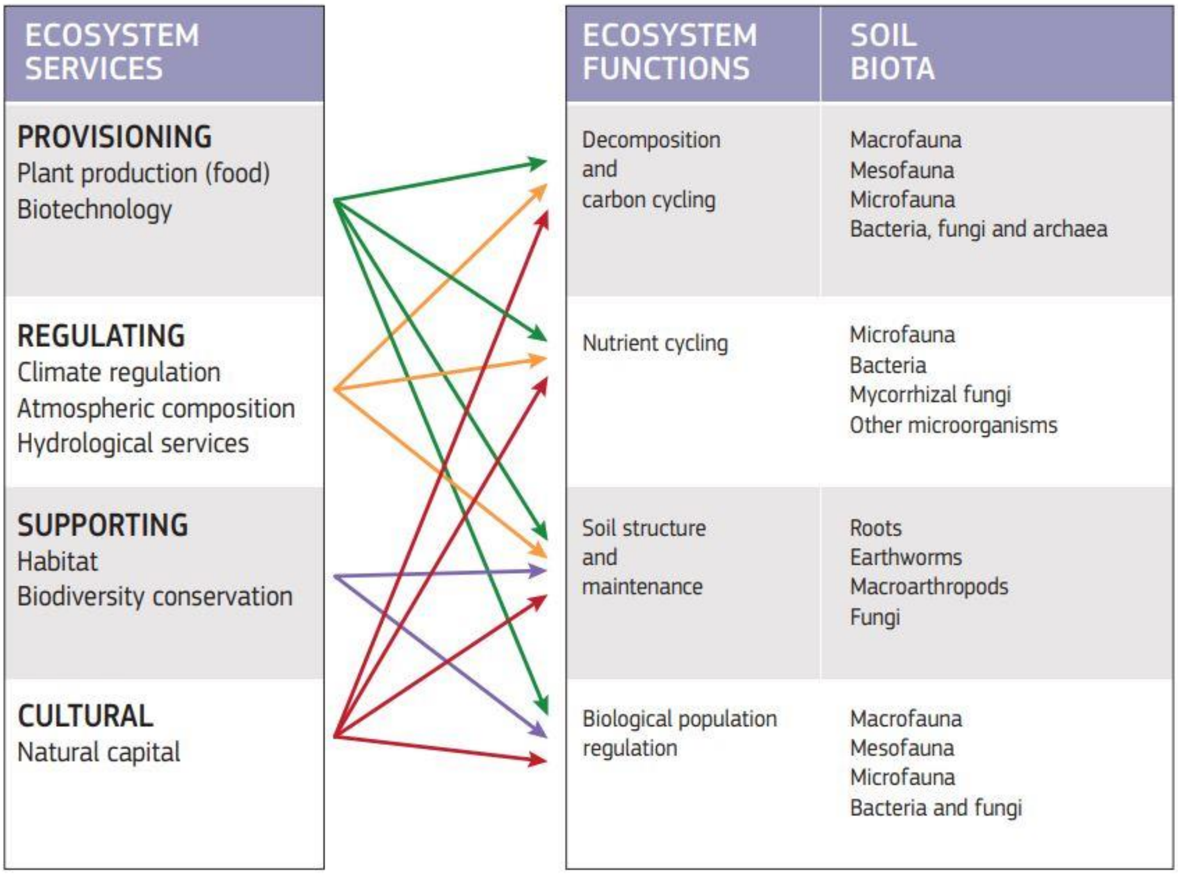


Figure 2-10. Soil-based ecosystem services, ecosystem functions and soil organisms that support them, from (Orgiazzi et al., 2016).

2.4 Soil degradation and threats

Just as ecosystem services are influenced by (bundles of) soil processes (i.e. aggregate soil or ecosystem functions), the latter are in turn affected by soil threats (Bünemann et al., 2018). The European Soil Thematic Strategy identified the main threats to soil quality in Europe as soil erosion, organic matter decline, contamination, sealing, compaction, soil biodiversity loss, salinization, flooding and landslides (EC, 2006). Although the economic value of these ecosystem services is difficult to calculate, it has been estimated that the consequences of soil biodiversity mismanagement are in excess of one trillion dollars per year worldwide (FAO et al., 2020; Orgiazzi et al., 2016; Turbé et al., 2010). Building on the conceptual framework established by Kibblewhite et al. (2008), Bünemann et al. (2018) more concretely link the abovementioned soil threats to specific soil functions and consequently impacted soil-based ecosystem services, see Figure 2-11 (shown also as 'SFD threats' in Figure 2-4).

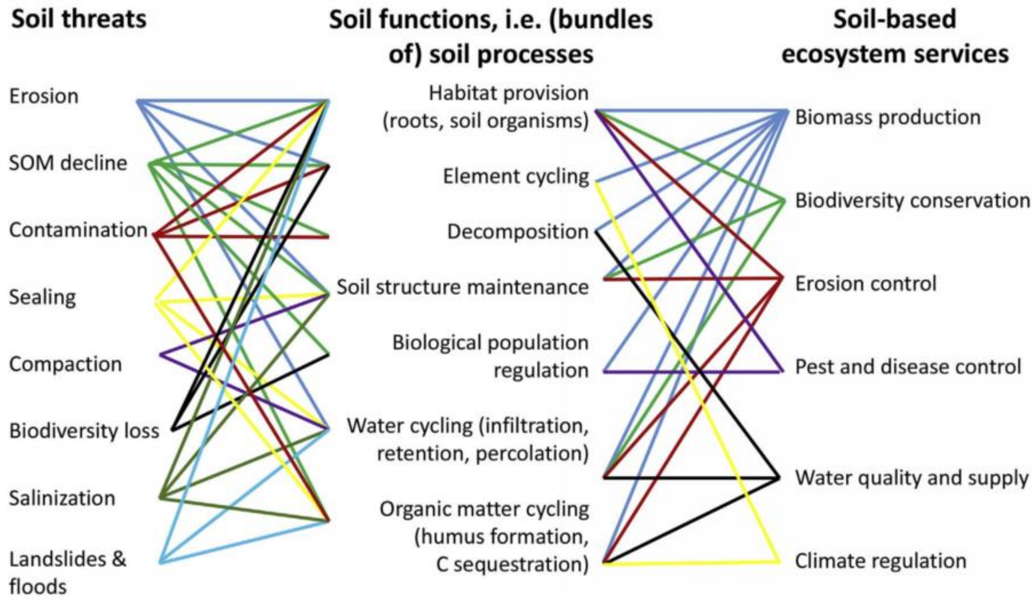


Figure 2-11. Linkages between soil threats, soil functions and soil-based ecosystem services. Further developed from the scheme presented by Kibblewhite et al. (Kibblewhite et al., 2008), *from* (Bünemann et al., 2018).

Wall et al. (2015) also made a clear connection between soil biodiversity, with its inherent complexity (the types, sizes, traits and functions of soil organisms), and human health since it provides ecosystem services like disease control and influences the quantity and quality of the food we eat, the air we breathe and the water we drink. Additionally, the authors stress that to achieve the Sustainable Development Goals it is not enough to aim towards improvement of a single benefit related to 'food' or 'air' or 'water' or 'disease' control, because all are simultaneously dependent on soils and soil biodiversity. Their assessment is rooted in the concept of human health as defined by The World Health Organization and Convention on Biological Diversity, which extends beyond disease and infirmity and recognizes human connections to other species, ecosystems and the ecological foundation of varied drivers and protectors of human health. The authors propose that promoting ecosystem functioning by managing soil biodiversity is a more cost-effective and sustainable approach than other resource-intensive methods to achieving long-term environmental and human health goals (Wall et al., 2015), see Figure 2-12. They further state that soil biodiversity is often negatively affected by the interaction between poor land management practices and drivers of climate change, both of which ultimately compromise ecosystem function and services that are essential for human health (e.g. control of pests, pathogens, production of nutritious food, cleansing water and reducing air pollution). Responses to reduced human health can in turn affect management decisions that govern land use and climate change (Wall et al., 2015).

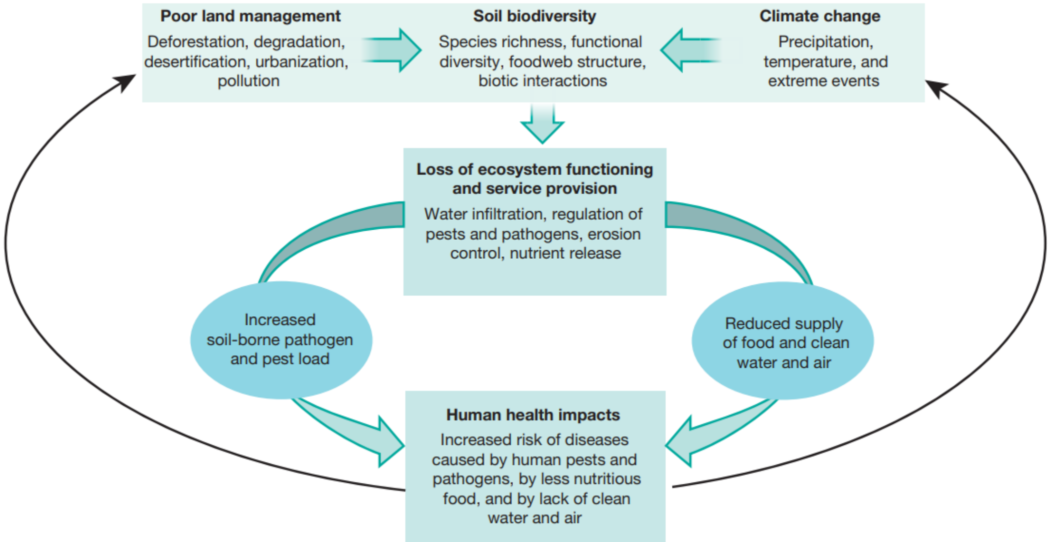


Figure 2-12. Flow diagram illustrating the link between soil biodiversity and human health, from (Wall et al., 2015).

Regarding specifically contaminated sites, the microbial community structure in soil can be markedly affected by chemical pollution, jeopardizing the provision of essential ecosystem services; thus, it is important to verify that during remediation processes, the links between soil biodiversity and soil functioning (as well as the corresponding ecosystem services) are restored (Gómez-Sagasti et al., 2012). Schröder et al. (2018) also make the case for intensifying biomass production on marginal land (a broader umbrella term that includes contaminated sites) for greater productivity as well as to address the degradation of soils by better managing this non-renewable resource. The authors discuss how valuable agricultural land has become abandoned due to contamination, and such sites will remain unproductive (and continue to pose ecological and human health risks) unless alternative strategies like gentle remediation options (e.g. phytoremediation and soil amendments) are used to reverse course and restore soil quality (Schröder et al., 2018). They emphatically state the following (exact wording as in source, (Schröder et al., 2018) pp. 1119):

> "From an ecological point of view, the rationale for restoration of degraded or marginal land is to recover lost aspects of local biodiversity and ecosystem resilience. From a pragmatic point of view, it is indispensable to recover or repair ecosystems and their capacity to provide a broad array of services and products upon which human economies and human life quality depends. For sure, it is a loss of culture and a loss of patrimony if we decide to abandon agriculture in an area" (Schröder et al., 2018).

3 Soil quality assessment

In recent years, due to the growing concern worldwide over the degradation of our soils, soil quality monitoring programs (including biodiversity) have been established in many countries (see (Pulleman et al., 2012) for an overview of European approaches focusing on soil biodiversity), and stress the importance of accounting for the soil biota in soil quality assessment. However, most of these monitoring programs comprise rather long lists of theoretically relevant (often very specific) indicators to be measured at different sites (according to soil types and land uses) and times even though no general agreement has been reached on their interpretation or direct linkage from land management to soil functions and ecosystem services (Baveye et al., 2016; Gómez-Sagasti et al., 2012; Pulleman et al., 2012; Velasquez et al., 2007).

A few prominent examples of methodologies for soil quality (or health, function or services) assessment include the following (see (Baveye et al., 2016; Bünemann et al., 2018; Turbé et al., 2010) for more extensive reviews):

Andrews et al. (2004) proposed a soil management assessment framework (SMAF), to evaluate soil quality, consisting of three main steps: indicator selection, indicator interpretation and integration into a soil quality index. Indicator selection was broken down into a minimum data set (MDS) of quantitative soil quality indicators (SQI) sensitive to changes in soil function. This method is considered unique in that there is a degree of flexibility in the customisation of the MDS and indicator selection step. These SQI were then divided into categories corresponding to a set of soil functions (biodiversity and habitat, filtering and buffering, nutrient cycling, physical stability and support, resistance and resilience, and water relation) including biological, physical and chemical parameters, which could be integrated into a scoring index (Andrews et al., 2004).

Velasquez et al. (2007) proposed a general Indicator of Soil Quality (GISQ) that evaluates soil ecosystem services (assuming that the more ecosystem services produced, the better is the soil quality) through a set of five sub-indicators: physical quality (e.g. soil hydraulic properties like soil porosity and compaction), chemical fertility (e.g. nutrient availability and pH), morphology (e.g. aggregate stability of the upper 5 cm, relating to hydraulic properties and C sequestration), organic matter stocks (e.g. climate regulation via carbon storage, soil fertility and as an energy source for biological activity) and macrofauna biodiversity (e.g. macroinvertebrate composition and abundance). To this end, 54 properties commonly used to describe the multifaceted aspects of soil quality were assembled into the aforementioned five sub-indicators and combined into a single numerical GISQ (Velasquez et al., 2007).

The Cornell Soil Health Test (Gugino et al., 2009; Moebius-Clune et al., 2016; Schindelbeck et al., 2008) is a standardised methodology assessing soil quality by integrating the physical, biological and chemical aspects of agricultural soils (also available online: Comprehensive Assessment of Soil Health (cornell.edu)). Initially evaluating 39 soil health indicators, 4 physical, 4 biological and 7 chemical indicators were selected for inclusion in the test based on sensitivity to management, relevance to functional soil processes, ease and cost of sampling and cost of analysis. The soil health test is targeted directly at land users, offering various soil health testing packages for farmers, landscape managers and others, and supplying them with management advice together with the results.

Garbisu et al. (2011) proposed evaluating soil quality according to the ecosystem health concept, i.e. as a measure of a system's *vigour* (productivity, throughput of material and energy in the system), *organisation* (diversity, of components and their degree of mutual dependence), *stability* (resilience and resistance, system's ability to maintain its structure and behaviour in the presence of stress), *suppressiveness* (disease resistance, severity or incidence remains low), and *redundancy* (functional, not affected by loss of a species if other species can perform the same functions). They maintain that soil microbial properties and parameters are the most relevant indicators/endpoints from an 'ecosystem health' perspective and propose functional groupings of microbial properties within the higher-level ecosystem health categories (or even ecosystem services) as a means of promoting greater understanding and communication, see Figure 3-1. An operative definition of soil quality nested within the concepts of ecosystem health is provided; where, *Soil quality is the capacity of a given soil to sustainably perform its ecological processes, functions and ecosystem services, and maintain a suite of essential ecosystem attributes of ecological relevance (vigour, organization, stability, suppressiveness, redundancy) at a level similar to that of a reference soil, without causing an adverse impact on the proper functioning of surrounding ecosystems or human health* (Garbisu et al., 2011). The assumption being made is that 'the higher the value of the ecosystem health attribute, the better the soil quality' appears to be valid for most situations and land uses (Garbisu et al., 2011). In later research, Epelde et al. (Epelde et al., 2014b) assessed soil quality by grouping select soil microbial parameters within the abovementioned ecosystem attributes (later expanded upon in Burges et al. (Burges et al., 2017, 2016) to group the parameters into ecosystem services). A soil quality index was then used to statistically calculate the overall soil quality using Equation (1) below:

$$Attribute\ (ES) = 10^{logm + \frac{\sum_{i=1}^{n}|logn_i - logm|}{n}} \qquad (1)$$

Where, *m* is the control value (set to 100%) and *n* corresponds to the measured values for each parameter as a percentage of the control value (Epelde et al., 2014b). Epelde et al. (2014) conclude that this index is appropriate for the assessment of soil quality in those cases where the soil has been intentionally treated to increase some parameters, e.g. addition of amendments or plants to remediate soil. More information on this methodology is covered in section 4.1.1 (pg. 74).

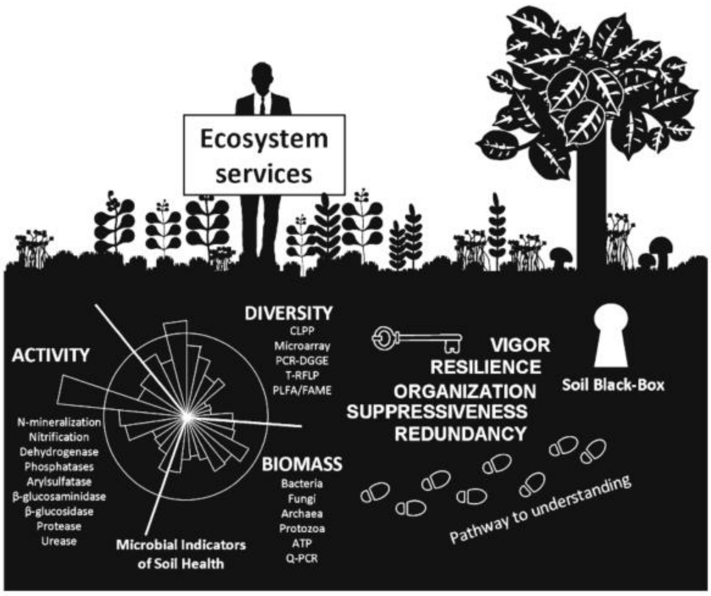


Figure 3-1. For a better interpretation of soil microbial properties as indicators of soil quality, it might be helpful to group microbial properties within a set of ecosystem health attributes of ecological relevance: vigour, organisation, resilience, suppressiveness, and redundancy, from (Gómez-Sagasti et al., 2012).

Faber and van Wensem (2012) and Thomsen et al. (2012) proposed grouping soil quality indicators within a set of ecosystem services in an attempt to enhance the impact of ecological risk assessment (ERA) on decision making. ERA for soils is often based upon the Triad approach (where, chemical, toxicological, and ecological data from a contaminated site are assessed along converging lines of evidence), and the authors propose to integrate ecosystem services as endpoints (i.e. measurable targets for protection or enhancement) within the ERA. The following ecosystem services were selected: soil fertility, adaptability and resilience, buffer and reaction function, biodiversity and habitat provision, disease suppression and pest resistance, and physical structure. To integrate ecosystem services within ERA, ecosystem services are first broken down into ecological requirements, i.e. the actual structures or processes of the ecosystem that underlie ecosystem services and which ecosystem provision is dependent upon (e.g., functional and structural biodiversity, ecosystem productivity, natural attenuation, and organic matter mineralization). The authors maintain that soil ecological integrity is what provides biological productive land and there is a need for at least a minimum appreciation of soil ecosystem requirements to the soil microenvironment quality. Physical, chemical or biological indicators are selected to assess the state of the ecological requirements. Ecosystem services, their ecological requirements, and indicators can be ranked or weighed by either societal or ecological importance with respect to a specific type of land use, affecting weighing of assessment results and thus the outcome of the decision-making (Faber, 2006; Faber and Van Wensem, 2012; Thomsen et al., 2012). More information on this methodology is covered in section 4.1.1 (pg. 74).

Volchko et al. (2014, 2019) created the Soil Function (SF) Box tool to evaluate the effects on soil functions (e.g. as the basis for primary productivity) in remediation projects as a key criteria to determine the best remediation option for contaminated brownfields. The updated version of SF Box (Volchko et al., 2019) is aimed to improve the basis for ecological risk assessment (ERA) by seeking to answer the questions "*what can this soil actually do and can it perform its Life and Habitat function well, assuming that it is free of contaminants?*" By addressing these questions, it can assist in making a distinction between the effects of contamination on soil biota and the effects of soil capability to function as a habitat to these species in a reference state free of contaminants. In the tool, a minimum data set of physical (soil texture, content of coarse material, available water capacity), biological (organic matter content, potentially mineralizable nitrogen) and chemical (pH, available phosphorous) soil quality indicators that can be used to evaluate the effects on soil are scored and integrated according to a soil quality index. Figure 3-6 depicts the generic framework for soil function assessment in a stepwise procedure using soil quality indicators (Volchko et al., 2014a, 2019).

3.1 Soil quality indicators

As shown above, there have been many attempts to create a comprehensive method for evaluating soil quality (or soil health) using *indicators* (alternatively called metrics, surrogates, proxies, etc.) – a term referring to both the parameter of interest as well as the corresponding method for evaluation, since both pieces are necessary for scientific comparability and consistency (Ritz et al., 2009). The aim is to identify and measure biotic or abiotic characteristics that are correlated (or at least thought to be) with soil functions and services of interest (Baveye et al., 2016). Which indicators to use assessing which aspects of the soil in what methodology (and accompanying terminology) has been a matter of debate for many years (e.g. (Doran and Zeiss, 2000; Karlen et al., 2003, 1997)). Clearly, no single indicator will encompass all aspects of soil quality, nor would it be feasible (or necessary) to measure all possible indicators (Kibblewhite et al., 2008). It is widely recognised that an important component of soil quality assessment is the identification of a set of sensitive soil attributes that reflect the capacity of a soil to function and can be used as indicators of soil quality (Bünemann et al., 2018). The selection of potential biological indicators is, however, only a step in developing practical soil quality assessment procedures (Doran and Zeiss, 2000), as there are operational issues to be solved. Selection criteria addressing these issues have commonly been applied to filter the extensive range of potential bioindicators (e.g. (Bünemann et al., 2018; Doran and Zeiss, 2000; Faber et al., 2013; Griffiths et al., 2016; Stone et al., 2016b; Turbé et al., 2010)). Considering more than only the technically-focused assessment (i.e. based upon how easy or inexpensive a specific method is to use) provides a greater robustness to the indicator selection and ranking process (Faber et al., 2013; Stone et al., 2016b) (discussed further in the following sections). Generally speaking, the indicators used to assess soil quality should meet the following criteria, from (Turbé et al., 2010) (see (Bünemann et al., 2018) for a review of criteria used in the selection of SQI):

1. **Meaningfulness** – well-correlated with beneficial, important soil functions that can provide valuable information using good surrogates (e.g. recognised high value organisms in functional groups).
2. **Standardisation** – parameters should be standardised (or at least readily available) to ensure comparability of data.
3. **Measurability and cost-efficiency** – parameters should be easy, inexpensive and accessible not only by experts, in order to ensure that the indicators will be used in practice and can be routinely collected. Also, relates to the availability of the necessary laboratory equipment and technical skills as well as labour-intensiveness in the field and lab (Faber et al., 2013).
4. **Sensitivity/Accuracy** – indicators should reflect changes and variations in management, land use or disturbances like contamination.
5. **Understandability** – indicators should be simple, easily understood and useful to land managers in decision-support (i.e. *fit for use* (Faber et al., 2013)).
6. **Policy-relevance** – indicators should be sensitive to sensitive to changes at policy-relevant spatio-temporal scales and allow for comparisons with a baseline situation to capture progress towards policy targets (e.g. concerning biodiversity and ecosystem function).
7. **Spatio-temporal coverage** – indicators should be validated in a wide range of conditions and should be amenable to aggregation or disaggregation at different spatial scales, from ecosystem to national and international levels.

In addition, as noted by Dickinson et al (2005), the presence of soil organisms provides the most obvious visual indicator of soil health (quality) but surprisingly often even this is not a standard item of soil quality evaluations. The authors state that there are justifiable reasons for this, complications which must be addressed include: (i) limited agreement on what organism or groups of organisms are most appropriate, (ii) high-level taxonomic skills required to assess groups of invertebrates, (iii) specialist equipment or approaches required in soil microbiology, (iv) the immense choice of potential indicators, and (v) lack of universal applicability for indicators of soil quality (Dickinson et al., 2005). They further state that for an SQI to be considered for use at contaminated sites, the functional group should be dominant in all soil types having a high biodiversity and abundance, have a significant role in the food web and be both sensitive to contamination and well-correlated with beneficial soil functions (Dickinson et al., 2005).

No single indicator will comply with all these criteria. In practice, efforts has been placed on the development of sets of complementary indicators, including both biotic and abiotic parameters, as selected by users (Pulleman et al., 2012; Turbé et al., 2010). However, despite the fact that a multitude of indicators estimating some aspect of soil biodiversity exists, no reference set of standardized indicators is available (Doran and Zeiss, 2000; Pulleman et al., 2012; Turbé et al., 2010). Instead, there are many different sets of proposed indicators to pick and choose from (see (Bünemann et al., 2018; Pulleman et al., 2012; Turbé et al., 2010) for reviews of these proposals).

The indicator-based approach is not without criticism, and it has been argued that the correlations between the indicators and relevant functions is not based on actual data but on expert opinion (e.g. (Baveye et al., 2016)). Indeed, the link between biodiversity and ecosystem services is sometimes believed to be so strong that is often considered implicitly that changes in biodiversity automatically correlate with changes in ecosystem services, but this link is not at all certain (Baveye et al., 2016). Kibblewhite et al. (2008) label this approach of using soil quality indicators to assess specific soil properties as 'reductionist'; describing it as an accessible and practical means of assessing soil condition, but potentially losing sight of the complex interactions (e.g. assemblages) making up the soil as an integrated, living system.

The soil system is undoubtedly complex and unlikely to be directly measurable, so for practical purposes surrogates (e.g. status of the soil biota) must be sought (Baveye et al., 2016; Kibblewhite et al., 2008). It has been argued by several authors that soil quality can only really be assessed in relation to one or several soil functions (e.g. Figure 3-2), ecosystem services or soil threats ('fitness for use') (e.g. (Baveye et al., 2016; Bünemann et al., 2018; Kibblewhite et al., 2008; Thomsen et al., 2012; Volchko et al., 2013, 2014a)). Therefore, according to Bünemann et al. (2018), clear definitions of these terms as well as firmly established associations with soil quality indicators are the basis of any functional soil quality concept (hence, the Terminology section).

Soil Quality Indicator	Soil Function				
	Sustain biological diversity, activity, and productivity "D"	Regulate and partition water and solute flow "W"	Filter, buffer, degrade, detoxify organic and inorganic materials "F"	Store and cycle nutrients and carbon "N"	Physical stability and support for plants and structures associated with human habitation "S"
Aggregate Stability [a,c,f]	★★	★★	—	★★	★★½
Available Water Capacity [a,g]	★★★	★★½	—	★★	—
Bulk Density [a,h]	★★½	★★★	—	★	★★★
Earthworms [b,d]	★★★	—	★★★	★★★	★★★
Infiltration [b,e,i]	—	★★	★	—	—
Particulate Organic Matter [a,c]	★★★	★★★	★★★	★★★	★★★
Potentially Mineralizable Nitrogen [a,c]	★★★	—	—	★★★	—
Reactive Carbon [a]	★★	★	★★★	★★	★★
Slaking [b,e,i,j]	★	★★½	—	—	—
Soil Crusts [b,d]	—	★★½	—	—	—
Soil Electrical Conductivity [b]	—	★★★	—	—	—
Soil Enzymes [a]	★★★	—	—	★★★	—
Soil Nitrate [b]	★	★	—	—	—
Soil pH [b,d]	★★	★★★	★★★	★★★	—
Soil Respiration [a,b,c]	★★★	—	★	★★★	★★
Soil Structure and Macropores [b,d]	★½	★½	★	★	★★
Total Organic Carbon [a]	★★★	★★★	★★★	★★★	★★★

[a] laboratory/office method
[b] field method
[c] time consuming
[d] simple visual observation
[e] variability requires large sample number
[f] perhaps the most informative physical indicator
[g] important for drought prone areas
[h] important for weight to volume conversions, small sampling errors result in significant interpretation problems
[i] effective educational method
[j] qualitative

Figure 3-2. Soil function – indicator matrix: when a direct relationship exists between the function and indicator, increasing reliability and ease of use of the associated assessment method is shown with increasing stars, from (USDA Natural Resource Conservation Service, 2015).

Though terminology may vary, soil quality and soil quality indicators have become internationally accepted, robust science-based tools for assessing soil resources (Karlen et al., 2003). The soil system is undoubtedly complex, and the soil quality concept acknowledges that soils have both inherent and dynamic properties and processes, and that soil quality assessment must account for these biological, chemical and physical properties and processes by using appropriate soil quality indicators (Karlen et al., 2003). The following sections will go more in-depth into specific aspects of the soil system that can be measured using the many and varied soil quality indicators.

3.1.1 Biological indicators

According to the previously established definitions, soil quality typically relates to a more human-centred evaluation of the soil properties and processes necessary to fulfil functioning suited for an end use. The more ecologically minded term soil health conveys the idea of soil as a living system with due consideration paid to the vast array of living organisms that are ultimately responsible for many soil functions (e.g. nutrient cycling, maintaining soil structure, etc.). To date, assessment and monitoring of soil quality has focused mainly on physico-chemical soil properties as indicators (e.g. pH, organic matter content, CEC, nutrient availability, water capacity, soil texture, etc.), but biological parameters are becoming increasingly used in soil assessments as they can provide a direct measure of soil functioning (Alkorta et al., 2003; Bünemann et al., 2018; Epelde et al., 2009a; Faber et al., 2013; Garbisu et al., 2011; Gómez-Sagasti et al., 2012; Orgiazzi et al., 2016; Ritz et al., 2009). Biological parameters are advantageous because they 1) are considered to be more sensitive to changes in the soil (e.g. due to management practices or contamination), 2) respond rapidly to changes over a shorter time period (i.e. dynamic), 3) provide information pertinent to many environmental factors, and 4) integrate complex, multi-dimensional phenomena that more directly relate to ecological status and function that are directly influenced by soil biota (Alkorta et al., 2003; Bünemann et al., 2018; Epelde et al., 2009a; Kibblewhite et al., 2008; Ritz et al., 2009). Regarding the criteria mentioned previously, Doran and Zeiss (2000) argue that soil organism and biotic parameters (e.g. abundance, diversity, food web structure, or community stability) meet most of the criteria for useful indicators of soil quality. Thus, these biological (or ecological) indicators (i.e. *bioindicators)* can be used to assess the status and changes in ecological soil properties and processes within a given physico-chemical context, and ought to be considered in any soil quality monitoring program (Ritz et al., 2009).

In their analysis to select suitable bioindicators for soil monitoring, Stone et al. (2016b) divided the bioindicators into those pertaining to 'biodiversity' or 'ecosystem function,' see Table 3-1, as is done here. However, many of the biodiversity indicators were identified as also relevant to assessing ecosystem function, including earthworms, enchytraeids, mites, collembola, nematodes and protista diversity. Some of the most studied bioindicators are briefly discussed in the following sections, broadly divided into indicators assessing primarily biodiversity or ecological function.

Table 3-1. List of indicators processed by logical sieve and their importance to either biodiversity or ecosystem function, from (Stone et al., 2016b).

Indicator	Biodiversity	Function
Macro- and mesofaunal diversity		
Earthworms (morphological identification or molecular methods)	X	X
Enchytraeids (morphological identification or molecular methods)	X	X
Mites (morphological identification or molecular methods)	X	X
Collembola (morphological identification or molecular methods)	X	X
Nematodes (morphological identification or molecular methods)	X	X
Protista diversity (morphological identification or molecular methods)	X	X
Microfaunal diversity		
Bacteria and Archaea species by molecular methods	X	
Fungi species (morphological identification or molecular methods)	X	
Bacteria and Fungi diversity through fingerprinting methods (TRFLP, ARISA)	X	
Pyrosequencing of soil DNA	X	X
PLFA	X	
Ecosystem function performed by soil biology		
Functional genes (targeting antibiotic producers, nitrifiers, denitrifiers)		X
Chip technology (up or down regulation of specific genes)		X
Molecular microbial biomass		X
Respiration (all basal methods)		X
Respiration (SIR-Glucose)		X
Respiration (Multiple-SIR)		X
Respiration (Biolog)		X
Nitrification potential		X

Multiple enzyme assay		X
Bait lamina		X
Litter bags		X

Biodiversity

Biodiversity is a soil attribute in itself, and therefore implicit within the ecosystem approach (Doran and Zeiss, 2000). Soil biodiversity is a key component of soil quality and can provide valuable information on a variety of endpoints including the interconnection between soil organisms and soil functions (Creamer et al., 2016a; Pulleman et al., 2012; Stone et al., 2016b, 2016a). Soil biodiversity can be characterised at the individual and community level, but presents difficulties due to the sheer enormity of biodiversity in the soil system, the opacity ('black box') of the soil matrix and that only a minute fraction of the bacterial and archaeal soil microbes will grow in culture media (i.e. laboratory environments) (Wurst et al., 2013). Often, to contextualise biodiversity measurements, biodiversity indices will be used to statistically quantify species diversity and composition (e.g. Shannon's diversity index), species richness (e.g. Margalef diversity index) or evenness (e.g. Pielou diversity index) (Volchko, 2014). Methods for assessing biodiversity have advanced considerably in recent years and can be broadly split into 1) morphological/physiological identification and 2) molecular/genetic methods. Advances in technology have enabled the more sophisticated, multi-endpoint DNA-based, genetic methods to be more reliable and informative than older physiological methods (Barrios, 2007; Bünemann et al., 2018). They are becoming more commonly applied in soil monitoring programs; however, they are more complex, expensive and not yet widely available, (Faber et al., 2013; Griffiths et al., 2016; Ritz et al., 2009; Stone et al., 2016b). Wurst et al. (2013) argue that the recent advances in genetics and the various guises of '-omics' (genomics, etc.) are revolutionising soil biology and will play an increasingly larger role in characterising soil biodiversity as the technologies become more refined.

In their review, Bünemann et al. (2018) note that molecular methods focusing on DNA and RNA have great potential to perform faster, cheaper and more informative measurements of soil biota and soil processes than conventional methods. Consequently, they may yield so-called 'novel indicators' that could substitute or complement existing biological and biochemical soil quality indicators in regular monitoring programs when the performance and cost-efficiency is improved. The rapid evolution of these techniques and the decreasing costs associated with them will likely increase their favourability and facilitate this development. However, the practical operability of these indicators by different stakeholders needs to be taken into account as there exist many limitations that limit their practical application in routine soil quality assessments. For example, the absence of standard operating procedures and accepted threshold values, especially for molecular methods, make the comparison and the interpretation of the results challenging. Also, the lack of direct functional linkages with soil processes and management implications further limits their application and potential use by stakeholders. Despite their promise and utility, Bünemann et al. (2018) conclude that **most novel soil quality indicators still belong to the research domain,** and many technological, practical and interpretation related issues need to be overcome.

A non-exhaustive list of common bioindicators used to assess biodiversity are listed below (see (Bünemann et al., 2018; Creamer et al., 2016a; Garbisu et al., 2011; Gómez-Sagasti et al., 2012; Griffiths et al., 2016; Pulleman et al., 2012; Ritz et al., 2009; Stone et al., 2016b) for more in-depth reviews discussing the various biodiversity related indicators, Soil Quality for Environmental Health website for more information on indicators and functions and Appendix II for a table showing the connections between the groups of soil biota and soil functions and services):

1. **Microbial diversity** – including bacteria, fungi and archaea.

Physiological: community-level physiological profiles (e.g. Biolog™ plates or microarrays) – often relating to functional diversity; for example, with Biolog™ Ecoplates by assessing bacterial activity in the presence of various substrates relating to specific soil functions (see e.g. (Campbell et al., 2003, 1997; Rutgers et al., 2016) for more information on Biolog™ and its application).

Genetic: genetic profiles (e.g. PCR-denaturing gradient gel electrophoresis (DGGE), phospholipid fatty acids (PLFA) or high-throughput genetic sequencing fingerprinting methods like TRFLP or ARISA) – to determine species abundance and richness (e.g. of bacteria and fungi), DNA abundance and gene sequences specific to individual species.

2. **Microfauna** – primarily assessing nematodes.

   Physiological: feeding guild richness (i.e. identifying plant-feeders, fungal-feeders, omnivores, bacterial-feeders, predators), total abundance (e.g. biomass per m^2) or nematode maturity index (MI) to assess soil condition based on nematode development.

   Genetic: pyrosequencing and other molecular methods (see e.g. Römbke et al. (2018)).

3. **Mesofauna** – including mites, Collembola (springtails), and enchytraeids (potworms).

   Physiological: species abundance (e.g. biomass per m^2) and richness (e.g. number of species per m^2).

   Genetic: pyrosequencing and other molecular methods (see e.g. Römbke et al. (2018)).

4. **Macrofauna** – including earthworms and other larger invertebrates.

   Physiological: species abundance (e.g. biomass per m^2) and richness (e.g. number of species per m^2).

   Genetic: pyrosequencing and other molecular methods (see e.g. Römbke et al. (2018))

   Note: Based on results from the French "Bioindicator" programme, earthworms are considered to be good environmental indicator candidates for sites contaminated with PAHs and metals since (i) they are well represented in the soil system in terms of density, (ii) they respond to a variety of environmental and ecological factors such as changes in soil chemistry, and forestry and agricultural practices, and (iii) they can be considered as an indicator of soil functioning due to their strong impact on soil (see Pérès et al. (2011) for more information).

Ecological function

Bioindicators pertaining to ecological functioning aim to directly assess the status of soil flora and fauna that are responsible for (or contribute to) soil processes and functioning (see Soil Quality for Environmental Health and Fact Sheets | soilquality.org.au websites for more information and fact sheets on indicators and functions). For example, soil microbial properties have received increasing interest as soil bioindicators due to their quick response, high sensitivity and direct ecological relevance to ecosystem functions (Epelde et al., 2009a, 2008; Garbisu et al., 2011; Gómez-Sagasti et al., 2012). Since microbial communities play key roles in many soil processes (e.g. nutrient cycling, organic matter decomposition) and the delivery of essential ecosystem services, they can provide a direct measure of soil functioning through assessing microbial biomass, activity and diversity (Gómez-Sagasti et al., 2012; Ritz et al., 2009). A non-exhaustive selection of common bioindicators used to assess ecological function are listed below, where #1-5 pertain to microbial indicators and #6 is used as a more aggregated indicator of soil function across faunal groups:

1. **Soil microbial biomass (SMB)** – is a measurement of the mass of intact microbial cells in a given soil, usually estimated from the measurement of carbon or nitrogen content of these cells (Carson, n.d.; ISO, 1997a, 1997b). Microbial biomass is an important constituent of the soil biological fertility, involved in the biogeochemical cycle of nutrients and carbon, and is an important reservoir of nutrients in ecosystems (Niemeyer et al., 2012). Soil microorganisms immobilize carbon and nitrogen by forming new biomass using the energy they obtain from oxidation of carbon sources through respiration, or inorganic chemical reactions (Gonzalez-Quiñones et al., 2011; Niemeyer et al., 2012). Therefore, more microbial biomass can stock, cycle and slowly release more nutrients, thereby improving the sustainability of an ecosystem (Gonzalez-Quiñones et al., 2011; Islam and Wright, 2004; Niemeyer et al., 2012). SMB is a critical component of the soil ecosystem that regulates many critical functions including nutrient cycling, decomposition of organic residues, structural stability and functioning as an indicator of soil pollution and bioremediation amongst others (Gonzalez-Quiñones et al., 2011; Islam and Wright, 2004; Niemeyer et al., 2012). Consequently, sites with low microbial biomass can have these functions impaired.

   Soil microbial biomass carbon (MBC) is the most commonly measured (by e.g. a fumigation-extraction method, ISO 14240-2:2011, or estimated from substrate-induced respiration, ISO 14240-2:2011) and is often used in a measurement to determine the 'metabolic quotient' (qCO_2), which is a ratio between basal soil respiration and MBC. This value reflects 'microbial efficiency' and microbial responses to stress, where a higher value indicates higher microbial stress, and has been used as an indicator of microbial stress caused by contamination in soil (Kumpiene et al., 2009; Niemeyer et al., 2012). MBC is the most commonly used parameter for the evaluation of soil microbial abundance and is a well-documented indicator of soil quality and fertility that has been shown to be sensitive to changes in soil management and disturbances (e.g. contamination) (Gómez-Sagasti et al., 2012; Niemeyer et al., 2012). In terms of interpreting the SMB values, absolute values are difficult to interpret so target or reference values are needed for soil quality assessments to allow ameliorative action to be taken at an appropriate time (Gonzalez-Quiñones et al., 2011)).

   Note: Broos et al. (2007) question the utility of routine measurements of MBC as an ecotoxicological endpoint at the field scale due to high spatial and temporal variability.

2. **Potentially mineralizable nitrogen (PMN), nitrification and ammonification rates –** are varying measurements of the biological activity of the soil with respect to nitrifying/denitrifying bacteria. PMN is an indicator of the capacity of the soil microbial community to convert (mineralise) nitrogen tied up in complex organic residues into the plant available form of ammonium (ISO, 2012a; Moebius-Clune et al., 2016). Various methods exist to measure the rates, most often utilising an incubation of a soil sample over a short (e.g. hours) or long (e.g. 7, 14 or 28 days) period of time then measuring the changes in quantities of respective nitrogen compounds (e.g. (ISO, 2012a, 2013; Moebius-Clune et al., 2016; OECD, 2000a)).

   Microorganisms in the soil are mainly responsible for nutrient cycling and nitrification/ammonification rates are key indicators with which to measure nitrogen cycling, and soil fertility, in soils (ISO, 2012a; Niemeyer et al., 2012). Nitrification is also considered to be one of the most sensitive soil microbial processes regarding contaminant-induced stress (Broos et al., 2005; Niemeyer et al., 2012). For example, Broos et al. (2005) found the potential nitrification rate to be a more sensitive ecotoxicological endpoint than substrate-induced and basal respiration or plant growth assays.

3. **Soil respiration** – methods for measuring respiration include two main types: 1) basal respiration (BR) – a value representing the baseline level of soil activity measured by O_2 uptake and/or CO_2 release without the addition of nutrients; and 2) substrate-induced respiration (SIR) – representing the 'potential' microbial activity by measuring the changes in O_2 uptake or CO_2 release after addition of various substrates like glucose (ISO, 2012b, 2002). Microbial soil respiration results from the mineralisation of organic substances and is a measure of the metabolic activity of the soil microbial community, where a higher CO_2 release indicates a larger, more active soil microbial community (ISO, 2002; Moebius-Clune et al., 2016). Methods like the Cornell Soil Health Assessment (Moebius-Clune et al., 2016) provide scoring functions to aid in interpretation of the absolute values obtained in such tests to evaluate soil heath. Otherwise, ratios like the 'respiratory activation quotient' (Q_R, a ratio of BR/SIR) or 'metabolic quotient' (qCO_2, a ratio of BR/MBC) to give indications of stress caused by e.g. contamination in a soil. More advanced SIR methods using multiple substrates (MSIR) like MicroResp™ have seen increasing use in recent years due to their better sophistication to gain a better understanding of the diversity of soil microbial community to utilise a wider range of carbon sources (e.g. (Campbell et al., 2003; Creamer et al., 2016b, 2009; Griffiths et al., 2016; Stone et al., 2016b)).

   Soil respiration is highly relevant to soil functioning as it is a direct measurement of biological activity, integrating abundance and activity of microbial life (Moebius-Clune et al., 2016). Thus, it is an indicator of the biological status of the soil community, which can give insight into the ability of the soil’s microbial community to accept and use residues or amendments, to mineralize and make nutrients available from them to plants and other organisms, to store nutrients and buffer their availability over time, and to develop good soil structure, among other important functions (Moebius-Clune et al., 2016). Furthermore, many studies have shown that soil respiration is a suitable soil quality indicator for comparisons between soil ecological conditions (e.g. contamination) and biological activity that responds well to gentle remediation options like phytoremediation (e.g. (Burges et al., 2016, 2017; Epelde et al., 2009a, 2010; Gómez-Sagasti et al., 2012; GREENLAND, 2014; Kumpiene et al., 2009)).

   From an ecotoxicity perspective, there is debate regarding the sensitivity of soil respiration, especially basal respiration (e.g. either increasing or decreasing in various assessments with increasing metal concentrations), to contamination (Broos et al., 2005; Niemeyer et al., 2012). For example, Broos et al. (2005) found that SIR (measuring the lag time between the addition of glucose and the exponential increase of the soil respiration rate) to be significantly more sensitive to metal toxicity than BR and more robust than other studied indicators in terms of reproducibility and consistency.

4. **Soil enzyme activity** – methods for measuring soil enzymes (though not the actual rate of enzymatic processes in-situ) in soil can involve such biochemical techniques as multi-well fluorometric assays that measure the fluorescence generated by soil samples diluted in buffer solutions contained fluorogenic substates corresponding to various soil enzymes (ISO, 2019a) or chemical extracts measured by spectrophotometry (ISO, 2019b, 2019c). Measuring soil enzymes provides a direct measurement of microbial activity as enzymes play key roles in the microbially-mediated processes of degradation and mineralisation of organic macromolecules, which is a crucial component of C, N, P and S nutrient cycling in soil (Table 3-2) (Alkorta et al., 2003; ISO, 2019a, 2019c, 2019b). Furthermore, such processes contribute to the decontamination of soil by degrading organic pollutants or immobilising heavy metals, participate in the formation of soil structure, and can have negative (plant pathogens) or positive (plant growth promoting rhizobacteria) effects on plant growth (Alkorta et al., 2003).

Table 3-2. Role of soil enzymes, adapted from Soil Quality: Indicators: Soil Enzymes.

Enzyme	Target organic substances	End product	Significance	Soil function
Beta glucosidase	carbon compounds	glucose (sugar)	energy for microorganisms	organic matter decomposition
FDA hydrolysis	organic matter	carbon and various nutrients	energy and nutrients for microorganisms, measure microbial biomass	organic matter decomposition, nutrient cycling
Amidase	carbon and nitrogen compounds	ammonium (NH4)	plant available NH4	nutrient cycling
Urease	nitrogen (urea)	ammonia (NH_3) and carbon dioxide (CO_2)	plant available NH4	nutrient cycling
Phosphatase	phosphorous	phosphate (PO4)	plant available P	nutrient cycling
Sulfatase	sulphur	sulphate (SO4)	plant available S	nutrient cycling

Soil enzymes are one of the more reactive components of the soil ecosystem (i.e. responding rapidly to changes in soil management and use) and potentially excellent soil quality indicators informing on the soil's microbial functional status and diversity (Alkorta et al., 2003; Epelde et al., 2008; Gómez-Sagasti et al., 2012; Touceda-González et al., 2017b). They are highly functionally relevant especially regarding nutrient cycling. Dehydrogenases indicate viable microbial activity and hydrolytic enzymes involved in key reactions in nutrient cycling (e.g. phosphatases, sulfatases, ureases, glucosidases, etc.) (Alkorta et al., 2003; Gómez-Sagasti et al., 2012). As bio-indicators soil enzyme activities show great value to monitor soil health, Alkorta et al. (2003) provide the following rationale for their use:

- They are related to important soil quality parameters (e.g. organic matters, soil physical properties, microbial activity, biomass),
- Change much sooner (i.e. *dynamic*) than other more static properties (e.g. soil organic carbon) in response to soil management practices or disturbances (i.e. *sensitive*),
- Serve as an integrative soil biological index of past soil management,
- Involve measuring procedures that are relatively simple and inexpensive.

The sensitivity of soil enzymes also makes them valuable indicators for contaminated soils. In the case of metals, which are toxic to living organisms primarily due to their protein-binding capacity and hence ability to inhibit enzymes, soil enzymes have been shown to be valid bio-indicators of the negative impact of heavy metals on the soil ecosystem as well as the effectiveness of gentle remediation options like phytoremediation (Epelde et al., 2008; Gómez-Sagasti et al., 2012; GREENLAND, 2014; Niemeyer et al., 2012). An important conclusion from the French "Bioindicator program" regarding soil enzymes are that enzymes are more sensitive to metallic contamination than to organic, enzymes are good indicators of metal bioavailability and that Alkaline phosphatase and Arylamidase are the most relevant enzymes to assess the effect of soil contamination (Cheviron et al., 2016).

5. **Functional genes** – measuring the abundance (copies/g soil or normalised per ng of DNA) of specific genes catalysing major transformation steps in nitrogen, carbon or phosphorous cycling, chemical transformations and plant growth promotion using techniques like quantitative-PCR assays. Functional genes for nitrogen cycling have been shown to be good candidate bioindicators (scoring highly in logical sieves – discussed in the following section) and highly sensitive to disturbances like contamination (Creamer et al., 2016a; Griffiths et al., 2016; Thiele-Bruhn et al., 2020; Tiberg et al., 2019; Volchko et al., 2020; Wessén and Hallin, 2011). Frequently assayed nitrogen cycling genes include:

   Denitrifying genes – *nosZ1* (clade I N_2O reducers), *nosZII* (clade II N_2O reducers), *nirK* (denitrifier type), *nirS* (denitrifier type) (Creamer 2016, Griffiths 2016, Volchko 2020)

Nitrifying genes – AOB (ammonia-oxidising bacteria) and AOA (ammonia-oxidising archaea) – *amoA* (total abundance) and *nifH* (total nitrogen-fixing bacteria) (Creamer 2016, Griffiths 2016, Volchko 2020)

6. **Bait lamina –** is a relatively simple, unobtrusive method for assessing the feeding activity of soil dwelling organisms (food webs both under and over the soil surface) *in-situ.* The method entails inserting small, perforated plastic strips enriched with organic material into the soil and counting the empty apertures of the bait-lamina strips after a certain exposure time (ISO, 2018). This method has a clear functional relevance by directly measuring feeding activity of soil fauna in particular (e.g. earthworms, termites, Collembola, enchytraeids) thus informing on organic matter decomposition (and nutrient cycling), functional integrity and the state of the soil as a habitat (Griffiths et al., 2016; ISO, 2018; Römbke, 2014). Bait lamina can also be used as a sensitive indicator to test the effects of disturbances like contamination (ISO, 2018).

   Note: Litter bags are similar tests to measure feeding activity through the loss of organic material *in-situ* but bait lamina is generally favoured due to ease of use (Griffiths et al., 2016; Römbke, 2014).

Summary

Compilations of the bioindicators listed above are provided in Table 3-3 and Table 3-4:

Table 3-3. Indicators of soil biodiversity according to functional group, adapted from (Turbé et al., 2010). *Highlighted cells indicate possible updated information, i.e. new ISO standards.*

Functional group	Organisms	Indicator	Method	Standard	Sensitivity to soil type	Sensitivity to land use	Measurability
Chemical engineers	Micro-organisms	Biomass/ activity	SIR, fumigation-extraction	Yes	Good	Good	Good
			ATP concentration, initial rate of mineralisation of glucose	Yes			
		Activity	Respiration rate/quotient ration	Yes	Good	Medium	Good
			Nitrification, N-mineralisation, C mineralisation	Yes	Medium	Medium	
			Denitrification	No	Medium	Medium	
			N-fixation	No	Good	Medium	
			Mycorrhizae (% of root colonised)	No	Good	Good	
		Enzymatic activity	Dehydrogenase activity	Yes	Good	Good	Medium
			Other enzymatic activity tests: phosphatase, sulphatase, etc.	No	Good	Good	Good
			Enzyme index	No	Very good	Very good	
		Diversity	Culture-dependent methods: direct count, community-level physiological profiles (CLPP)	No	Poor	Poor	Good
			Culture-independent methods: fatty acids analysis, nucleic acid analysis	No	Poor	Very good	Good (technical)
Biological regulators	Protists, nematodes	Abundance and diversity	Culture-dependent methods: direct count (diversity index, functional or trophic diversity)	Yes	Good	Very good	Low (time, expertise)
			Culture-independent methods: fatty acids analysis, nucleic acid analysis				
	Micro-arthropods (springtails, mites)	Counting	Litter-bag technique (colonisation capacity)	No	Good	Good	Low (time, expertise)
			Soil coring				
		Abundance and diversity	Community composition, ecological groupings	Yes	Very good	Very good	Low (time, expertise)
Soil ecosystem engineers	Earthworms, isopods	Abundance and diversity	Species richness, diversity, evenness	Yes (ongoing)	Very good	Good	Good (low expertise, simple)

Table 3-4. Soil biological indicators, methodologies, related main soil functions and advantages/disadvantages at different time scales, adapted from (Bünemann et al., 2018).

Indicator	Methodology	Main soil functions	Main pros	Main cons
Individual population and community level				
Presence, richness and abundance of individual soil organisms	Traditional handsorting and microscopic methods; molecular quantitation (qPCR)	Nutrient, organic matter and water cycling, biological population regulation, soil structure maintenance	Taxonomic and functional level	Not always linked directly with functions. Difficult to apply to fauna, e.g. protozoa, mites and collembola
Microbial biomass and fungal biomass, fungi:bacteria ratio	Direct counting, chloroform fumigation-extraction, SIR, PLFA, molecular quantitation	Nutrient and organic matter cycling, decomposition, soil structure maintenance	Sensitive and well related with other soil quality indicators	Spatially variable, difficult interpretation, contradictory results. Unclear direct link to functionality
Indices based on faunal communities (e.g. Maturity Index, Enrichment Index, Channel Index, Structural Index for nematodes)	Counting and identification of specific groups of organisms	Nutrient and organic matter cycling, biological population regulation, decomposition	Sensitive, taxonomic and functional level	Time-consuming and costly, specialist required for morphological identification
Community composition	Manual counting and identification	Nutrient and organic matter cycling, biological population regulation, decomposition, soil structure maintenance	Division in functional groups can give an indication of functions	Time-consuming, expertise required, not indicative of active biota
	PLFA		Correlated with other measurements, good indicator of active microbial biomass, integrated information on the microbial community	Time-consuming, no direct link with functions, coarse resolution
	Fingerprinting methods (e.g. DGGE, T-RFLP, ARISA, ARDRA, TGGE), microarrays		Greater phylogenetic resolution	No direct link with function, difficult comparison between studies due to greater variety in methods, difficulties to extract and amplify DNA
	Sequencing (metabarcoding)		Detailed view of diversity, enormous amounts of data, detects less abundant organisms, permits discovery of new diversity	Taxonomic genes with no direct link with functions, difficulties to extract and amplify DNA, costly, problems related with handling of large datasets and analyses, dependent on libraries, no standard methodology
	Community level physiological profiling (CLPP, e.g. Biolog™, MicroResp™)	Nutrient and organic matter cycling, decomposition, habitat provision	Insight into functionality of the community, MicroResp™ closer to in-situ conditions, shorter time of measurements	Many replicates needed because of variability

Ecosystem level				
Soil respiration, nitrogen mineralisation, denitrification, nitrification	CO_2 evolution, N_2O emission, NO_3 produced	Nutrient, organic matter and water cycling, decomposition, habitat provision	Sensitive and ecologically relevant	Highly variable and fluctuating, relatively laborious
Potentially mineralizable nitrogen	Anaerobic incubation		Good correlation with MB and total soil N	Relatively laborious
Metabolic quotient (qCO2), microbial quotient (MBC/SoilC)			Sensitive, simple and inexpensive	Difficult interpretation: confounds disturbance with stress
DNA and protein synthesis	Thymidine and leucine DNA incorporation		Reflection of active microbial biomass	No standardised procedure
Enzymatic activities	Extraction of enzymes in the soil and incubation with various substrates	Nutrient and organic matter cycling, decomposition, biological population regulation	Closely related to important soil quality parameters, very sensitive, simple and inexpensive methods	Standard procedure not available, contradictory results, complex behaviour and variable for each enzyme, potential activity
Functional genes and transcripts	FISH, Microarrays, *meta*-transcriptomic, qPCR, metagenome analysis		Closer link to functionality, FISH and microarrays can give an idea of active microorganisms, high sensitivity and throughput	Restricted to known gene sequences, genes and transcripts might not be expressed, difficulties linked with RNA extraction, costly
Metabolomics and metaproteomics	Assessment and quantitation of metabolites and proteins in the soil	Nutrient and organic matter cycling, decomposition, biological population regulation, soil structure maintenance	Closer link to functionality	Field in development, difficult extraction of metabolites and proteins
Stable isotope probing	Incorporation of 13C- or 15N-labelled substrates into DNA, RNA, PLFA, proteins	Nutrient and organic matter cycling, decomposition	Permits establishing link between biodiversity and functions, allows in-situ analysis of active microbial population	Field in development, time involved in the assimilation of the substrates

3.1.2 Indicator selection

Many bioindicators have been proposed for various purposes, and multiple reviews have been carried out to evaluate and filter them (e.g. by a 'logical sieve' (Ritz et al., 2009) or 'parameter selection module' (Gutiérrez et al., 2015)) according to multiple selection criteria, like those previously mentioned, to determine those best suited for soil monitoring programs for biodiversity and/or ecosystem function, e.g. (Faber et al., 2013; Griffiths et al., 2016; Ritz et al., 2009; Stone et al., 2016b). These guiding examples will be briefly discussed below:

In one of the first attempts, Ritz et al. (2009) presented a 'participatory approach' (enlisting the help of experts) of selecting soil biological indicators from a list of 183 potential indicators pertinent to a subset of three ecological soil functions (i.e. bundles of ecosystem process and properties):

- **Food and fibre production** – maintaining soil in a suitable state for plant and animal biomass production (supplying nutrients and water, disease control, physical condition)
- **Environmental interactions** – protecting the capacity of soils to store, transform and regulate soil processes (gas exchanges, degradation and retention of solid materials e.g. pollutants and organic water, water flow regulation) critical to environmental sustainability
- **Support of habitats and biodiversity** – maintaining the ecological, utilitarian and ethical value of soil biodiversity including maintenance of semi-natural habitats and biodiversity aboveground

To select for suitable indicators, the potential indicators were scored by scientists and end-users according to multiple technical and scientific criteria in a 'logical-sieve' approach, which allowed several iterations to account for end-user requirements and expert opinion. The different requirements for an indicator were weighted and, in combination with scores, were used to then rank, prioritise and select the indicators. A final ranked list of 21 indicators was produced that covered a range of genotypic-, phenotypic- and functional-based indicators for different trophic groups, though 4 of these were deemed not sufficiently robust for ready deployment, see Table 3-5. Genetic analyses predominate the results, partly reflecting advances in molecular techniques, since they relate directly to diversity and function (Ritz et al., 2009).

Table 3-5. Consolidated list of candidate indicators from the logical sieve by Ritz et al. (2009), *ranked according to aggregated score (F_A) and categorised according to deployment status (as of 2005). TRFLP = terminal restriction fragment length polymorphism, PLFA = phospholipid fatty acids, PCR = polymerase chain reaction.*

Indicator	Indicator Description	F_A	Sub-category
Deployment status = 2, F_A > 100			
TRFLP- ammonia oxidiser/denitrifiers	Genetic profile - specific group	769	Genotype - Nucleic acid
PLFA profiles	Composition - total community	615	Phenotype - Biomarker
TRFLP - ITS fungal	Genetic profile - specific group	437	Genotype - Nucleic acid
Multiple substrate-induced respiration (MSIR) GC	Activity capability profile - total community	311	Function - activity
Nematode Baermann extraction procedure	Numbers, composition and size of nematode community	302	Phenotype - Fauna
TRFLP - bacteria	Genetic profile - specific group	295	Genotype - Nucleic acid

Microarthropods Tullgren dry extraction	Numbers, composition and size of invertebrate community within soil	188	Phenotype - Fauna
On-site visual recording - flora and fauna	Numbers estimate of animals	173	Phenotype - Other
Microplate fluorometric assay - multi-enzyme	Enzyme potential activity - wide range	172	Function - Enzyme
TRFLP - Archaea	Genetic profile - specific group	146	Genotype - Nucleic acid
TRFLP - methanogens/methanotrophs	Genetic profile - specific group	123	Genotype - Nucleic acid
Invertebrates pitfall traps	Numbers, composition and size of invertebrates motile aboveground	123	Phenotype - Fauna
TRFLP - actinomycetes	Genetic profile - specific group	121	Genotype - Nucleic acid
Deployment status = 1, F_A > 100			
TRFLP - nematodes	Genetic profile - specific group	437	Genotype - Nucleic acid
Multiple substrate-induced respiration (MSIR) MicroResp	Activity capability profile - total community	313	Function - activity
TRFLP - protozoa	Genetic profile - specific group	291	Genotype - Nucleic acid
qPCR AM Fungi	Genetic profile - specific group	111	Genotype - Nucleic acid
Deployment status = 0, F_A > 50			
Functional gene arrays	Genetic profile	788	Genotype - Nucleic acid
Phylogenetic gene arrays	Genetic profile	511	Genotype - Nucleic acid
FISH - keystone species	Genetic profile	138	Genotype - Nucleic acid
Soil proteomics	Phenotypic profile	51	Phenotype - Other

Logical sieve(s) was also performed within the scope of the Ecological Function and Biodiversity Indicators in European Soils (EcoFINDERS[7]) project, carried out to determine and standardise the most valuable bioindicators to evaluate the relationships between soil biodiversity, ecosystem function and ecosystem services across various climatic zones, soil and land use types, see (Creamer et al., 2016a; Faber et al., 2013; Griffiths et al., 2016; Stone et al., 2016b, 2016a). The logical sieve was a modified version of the method proposed by Ritz et al. (2009) (discussed in broad detail in (Faber et al., 2013)) allowing customisation according to the desired end-use and enabling a structured ordering of potential indicators according to the following steps: 1) establishment of the purpose for which the monitoring will be applied (e.g. to monitor changes in soil biodiversity and ecosystem function across Europe); 2) listing of the potential indicators (consolidation into a shortlist for sieving – derived from surveys); 3) classification of indicators into operational categories (e.g. microbial, faunal, and functional); 4) ranking of indicators in order of their relevance to specific criteria. Indicators to sieve were shortlisted for relevance to ES (i.e. water retention, C sequestration, and nutrient provision). Faber et al. (2013) note that modifying the selection criteria used in the logical sieve approach is necessary to come up with an objectively filtered shortlist. The shortlist ought to be context-specific in terms of e.g. time and place, professional involvement, stakeholder preferences, budgetary restrictions and specific objectives and requirements of the monitoring network (Faber et al., 2013).

The full logical sieve was later applied, by building on the list of indicators identified in Faber et al. (2013) (Figure 3-3), and discussed in Stone et al. (2016b) to establish the most suitable bioindicators of soil quality for use in future soil monitoring programs across the agricultural

[7] https://projects.au.dk/ecofinders/

areas of Europe; concerning specifically the ecosystem functions of 1) habitat for soil biodiversity, 2) C cycling and storage, 3) cycling of nitrogen (N) and phosphorous (P).

The responses from surveys sent out to soil science experts were used to score each indicator (see Table 3-1 for list of indicators) according to the criteria and then rank them according to:

1. *Technical factor* (F_T) – gives an indication of the practicality associated with the measurement of a particular indicator (i.e. cost and difficulty). Comparing to the aggregate score, those that scored highest in this category were not necessary the best overall. The top ten indicators mostly determinant for ecosystem function (e.g. respiration, nitrification, molecular microbial biomass, litter bags) as they are usually simple and effective compared to high cost of molecular techniques and high labour demand of soil fauna identification.
2. *Applicability/Discrimination factor* (F_{AD}) – applicability tests the ubiquitous nature of each indicator (e.g. presence of earthworms at a location) and discrimination tests the sensitivity of indicators to environmental conditions. Five of the top ten indicators within this category were for ecological function (e.g. enzymes, nitrification, respiration, molecular microbial biomass, functional genes). **Indicators which measure ecosystem function are therefore intrinsically able to discriminate between these different soil conditions and are often recommended for monitoring schemes.** Molecular methods for biodiversity produce a lot of species data that can be used to create a detailed picture of a community or niche leading to higher discrimination ability, which can also be linked to functioning.
3. *Soil function factor* (F_{SF}) – changes the recommended indicators dependent on the function(s) wanted, thus function is a key parameter for selecting indicators suited to desired end use. The results can also change depending on whether the indicator is intended all functions or in a looser monitoring situation where a single function could be monitored by an indicator (e.g. potential nitrification). Molecular methods for microbial biodiversity score highest if ubiquitous (i.e. multiple) function measurement is desired. Important to note is that some indicators are highly specific and attributable to certain functions.
4. *Aggregate score* – combining the above factors into an aggregate score resulted in a mix of biodiversity and ecosystem functioning indicators due to: 1) the discrimination potential and wide applicability of biodiversity indicators and 2) a mix of technical factors and discrimination potential for the indicators of soil ecosystem function provision. The top ten indicators were dominated by molecular methods of measuring both biodiversity and ecological function (7 indicator/method combinations).

According to Stone, Ritz et al. (2016): '*Though methods of determining biodiversity are often time consuming and/or costly (with regard to scientist hours need for morphological identification), it can be seen from the results of this exercise that the potential of these indicators in terms of discrimination and relevance to more than one function results in them being recommended for use as indicators for monitoring schemes*.' This includes bacteria and archaea diversity, fungi diversity and mite biodiversity which all scored in the top ten of the aggregated scoring. Regarding ecosystem function, four indicators were in the top ten: 1) respiration (MSIR), 2) molecular microbial biomass, 3) functional genes (targeting antibiotic producers, nitrifiers and denitrifiers), and 4) multiple enzyme assays. Of these functional indicators, the latter two indicators scored higher than the former two indicators due to their relevance to more than one function.

Also within the scope of the EcoFINDERS project, Griffiths et al. (2016) performed a logical sieve to select cost-effective and policy-relevant bioindicators for monitoring of soil biodiversity and ecosystem function in Europe; concerning specifically ecosystem functions related to the ecosystem services of: 1) water regulation, 2) C-sequestration, and 3) nutrient provision. Their sieving approach was focused on the indicator selection process itself (a 'top-down' approach) with particular attention paid to the cost-effectiveness of the indicators (i.e. operation in the field, laboratory and equipment/instrumentation) and interpretation of the results from the monitoring. To that end, a shortlist of 30 potential bioindicators, see Table 3-6 below, was developed by a panel of experts then rigorously sieved following an approach like those mentioned previously. Eighteen of these sieved indicators (a 'logically feasible number for evaluation') were then tested in the field, ranked and grouped within 5 categories: 1) ease of field sampling, 2) utility (in terms of getting more than one piece of information from the test), 3) ease of lab test, 4) lab throughput, and 5) setup costs.

Table 3-6. Weighted score from the logical sieve assessment of potential biological indicators of soil biodiversity and ecosystem function. Indicators were grouped as faunal, microbial or functional, and addressed issues of biodiversity (BD) and/or ecosystem function (EF). Indicators selected for evaluation in the field are marked in bold. DNA abundance and resilience were not assessed in the logical sieve (marked n/a). EEA = extra-cellular enzyme activity; T-RFLP = terminal restriction length polymorphism of archaea, bacteria and fungi; PLFA = phospholipid fatty acids. From (Griffiths et al., 2016).

Potential indicator	Indicator group	Issue addressed	Weighted score
Nematodes: molecular	**Fauna**	**BD/EF**	**659**
Nematodes: morphological	Fauna		640
Enchytraeids: molecular	Fauna		639
Mites: molecular	Fauna		639
Collembola: molecular	Fauna		639
Earthworms: morphological	**Fauna**	**BD/EF**	**633**
Collembola: morphological	**Fauna**	**BD/EF**	**623**
Enchytraeids: morphological	**Fauna**	**BD/EF**	**623**
Mites: morphological	**Fauna**	**BD/EF**	**611**
Earthworms: molecular	Fauna		599
Fungi (ergosterol)	**Microbe**	**BD/EF**	**549**
Protista: molecular	Microbe		539
Nitrification	**Function**	**EF**	**525**
Potentially mineralizable N	**Function**	**EF**	**525**
Hot water extractable C	**Function**	**EF**	**525**
Respiration	Function	EF	507
Bait lamina	**Function**	**EF**	**492**
EEA	**Function**	**EF**	**474**
Microbial T-RFLP	**Microbe**	**BD**	**473**
PLFA	**Microbe**	**BD**	**459**
Functional genes	**Function**	**BD/EF**	**448**
Protista: morphological	Microbe		446
Denitrification	Function		422
Pyrosequencing	Microbe		415
MicroResp	**Function**	**EF**	**398**
Water infiltration	**Function**	**EF**	**398**
Molecular Chip technology	Microbe		383
Other 'omic' methods	Microbe		328
DNA abundance	**Microbe**	**EF**	**n/a**
Resilience	**Microbe**	**EF**	**n/a**

In their breakdown of the cost effectiveness of the indicators, results showed the expected trade-off between the intensity of work in the field and intensity in the laboratory (Griffiths et al., 2016). Thus, earthworms and water infiltration, which are labour intensive in the field, require relatively little laboratory time, while DNA based analyses from the easily obtained composite soil sample require the most laboratory effort. The authors further state that a monitoring programme should be based upon a suite of different indicators (since none could detect all management effects) to enhance reliability (Griffiths et al., 2016). However, this inevitably incurs a trade-off to balance between reliability (larger set) and costs (smaller set) when selecting indicators and designing monitoring system. They add that *'in any monitoring scheme there will be over-riding considerations of resources, time and expertise available, so any decision to apply extra tiers, further indicators or more complete datasets then becomes an internal matter that is different for each monitoring scheme*' (Griffiths et al., 2016). Furthermore, the authors emphasise that **standardisation is an absolute necessity to enable comparison of results**. General conclusions regarding methods can be separated into:

- **Molecular methods** – expensive and labour-intensive lab tests, the field is developing rapidly though results can be difficult to interpret, and new platforms (e.g. Illumina) could provide more information on molecular analysis of flora/fauna; however, many are not yet cost-effective, fully developed or standardised.
- **Non-molecular methods** – widely used throughout Europe in soil monitoring schemes and have undergone thorough scientific validation and clear demonstration of usefulness (e.g. score well in logical sieves) and cost-effectiveness.

Regarding biodiversity monitoring, as the presence of soil biota varies from site to site, the authors argue that all taxonomic groups would need to be included because changes in the biodiversity of one group cannot be used to infer changes in other taxonomic groups (Griffiths et al., 2016). The authors refer to the ENVASSO (ENVironmental ASsessment of Soil for mOnitoring) project ((Bispo et al., 2009; Faber et al., 2013)) which employs a 3-tiered approach aimed at defining and documenting a common soil monitoring system in support of a European Soil Framework Directive:

Tier 1 – a basic, minimum set of 3 indicators for large-scale adoption, standardised according to ISO with reference values for comparison in some cases. Indicators used are for species diversity (earthworms, collembolan species) and biological function (soil respiration)

Tier 2 – a more intensive study looking at the effects of a stressor, typically requires expert identification for additional biodiversity indicators (macrofauna, mites, nematodes, bacteria and fungi) and testing for bacterial and fungal activity. Such increased effort is only required when specific sites or questions must be addressed

Tier 3 – integrates the activity of the whole soil community, ecologically relevant but likely to be less sensitive in establishing effects. It calls for additional biodiversity (protist) and functional (faunal activity from litter bags or bait lamina) indicators but is optional and likely not directly linked to functioning.

For monitoring under the European climatic zones and land uses, Griffiths et al. (2016) suggest different indicators of ecosystem function than for monitoring of soil biodiversity. For ecosystem functions related to the services of water regulation, C-sequestration and nutrient provision (which are all carried out by the general soil biota), they would recommend a minimum data set from each indicator group:

- **Fauna:** *earthworms* – (measured in e.g. biomass g/m2) there could be too few species to be reliable indicator of biodiversity, however anecic earthworm species are strongly related to water infiltration and earthworms in general are important for many soil functions.

- **Microbial:** *functional genes* – (normalized gene copy number per ng DNA) genes for nitrogen cycling were selected as they have been shown to be good bioindicators for soil monitoring and are becoming increasingly common in scientific literature (e.g. (Wessén and Hallin, 2011)) and have also been shown to be sensitive to soil contamination ((Tiberg et al., 2019; Volchko et al., 2020)).
  Note: MicroResp and enzyme activities were also considered valid for use.
- **Functional:** *bait lamina* – constraints to using bait lamina sticks exist (e.g. cultivating too soon after deployment) but were shown to be easy to use, functionally relevant and sensitive to land use changes/management.

As demonstrated in these studies, bioindicators are necessary to gain a deeper understanding of the soil system and to monitor eventual changes that may result from land management strategies. Many Europe-wide sampling campaigns and projects utilising various bioindicators have been carried out to characterise and monitor soils (see (Pulleman et al., 2012; Turbé et al., 2010) for comprehensive reviews). These include the aforementioned ENVASSO and EcoFINDERS projects as well as other frameworks for selecting biological indicators for national soil monitoring have been devised in, for example, France (BioIndicator - (Pérès et al., 2011)) and the Netherlands (BISQ - (Rutgers et al., 2012, 2009)). These frameworks adopted a similar approach to the logical sieve; in which a wide range of candidate indicators were assembled and tested for their suitability to be used in systematic soil biodiversity assessment. Within the scope of EcoFINDERS, Stone et al. (2016a) established a range of sites representing a varied set of soils (a 'transect') across Europe to then be used for testing a calibrating methods of measuring soil biodiversity (e.g. micro- and mesofauna biodiversity, extracellular enzyme activity, phospholipid fatty acid and community level physiological profiling using MicroResp™ and Biolog™). Also within the scope of the EcoFINDERS project, Creamer et al. (2016a) performed an 'ecological network analysis' across this Europe-wide transect to greater understand the interconnections between soil biodiversity and ecosystem function (specifically carbon cycling and storage potential and nutrient cycling of nitrogen and phosphorous). They tested soils using biological indicators related to *microbial diversity*: DNA yields (molecular biomass), archaea, bacteria, total fungi and arbuscular mycorrhizal fungi; *microfauna diversity*: nematode trophic groups; *mesofauna diversity:* enchytraeids and Collembola species; and *microbial function:* nitrification, extracellular enzymes, multiple substrate-induced respiration, community level physiological profiling and ammonia oxidiser/nitrification functional genes. The network analysis was used to identify the key connections between organisms (and taxonomic units) under the different land use scenarios, reflecting their relative importance to soil functioning. *Key drivers of carbon cycling and storage over time were shown to relate to microbial biomass, basal respiration and fungal richness* while nutrient cycling showed more variation in key biological indicators across sites.

3.1.3 Minimum data set

As has been noted in the abovementioned studies, there are potentially hundreds of viable bioindicators that could be potentially used in a soil monitoring program (Faber et al., 2013; Griffiths et al., 2016; Ritz et al., 2009; Stone et al., 2016b). Hence, the selection of a minimum data set (MDS) derived from a larger set of soil quality indicators is a necessary step in soil quality assessments because of financial and time limitations (Bünemann et al., 2018). The logical sieve is advantageous due to its methodological transparency, which is imperative to allow wide application and replication of minimum dataset selection (Bünemann et al., 2018). According to Bünemann et al. (2018) **the number of indicators finally selected for inclusion**

in a minimum dataset typically ranges between 6-8, which could be viewed as the output of a logical sieve or other type of indicator selection process.

The assessment of soil health (quality) can be based on measures of biodiversity or functional processes, though Dickinson et al. (2005) argue that soil biodiversity is probably most important for maintaining ecosystem function in disturbed (e.g. contaminated) environments. They preliminarily suggest a test battery using bioindicators wherein biodiversity can be measured directly as species richness, or as a surrogate measure of biodiversity using standardized procedures (e.g. higher taxa richness, microbial community diversity, testate amoebae, nematode maturity indices), and functioning of soil processes can be measured as soil functional assessments (e.g. enzyme assays, nutrient mineralization, nitrification potential, soil respirometry) (Dickinson et al., 2005). In a follow-up paper, Hartley et al. (2008) attempt to establish a test battery with indicators selected due to their importance as key functional groups in soils. For example, earthworms as soil ecosystem engineers, microarthropods as microregulators (i.e. biological regulators) and microorganisms as decomposers and elemental transformers (i.e. chemical engineers). More than 30 potential assays were short-listed for investigation, with selection based on similar criteria to those mentioned previously. In their study, the most extreme sites (metal-contaminated or suffering from compaction) were readily identified by some of the assays, but performance of indicators varied considerably between sites (Hartley et al., 2008). They observed that the most successful site remediation, from the point of view of vegetation establishment (a landfill site), also scored well using assays of ATP, microbial carbon, soil respiration and microbial biodiversity (Hartley et al., 2008). When sites were ranked on the basis of an index that combined results from the more effective assays, there was a clear discrimination between degraded and less perturbed environments; however, the authors were forced to conclude: '*unfortunately, this could not be sustained by further critical analysis of data, leaving the conclusion that there is no obvious suite of robust or reliable indicators'* (Hartley et al., 2008).

Volchko et al. (2014) observed that there are a limited number of studies aimed at providing MDSs to assess soil quality for non-agricultural uses. A search in the Scopus database showed 2 hits for "brownfield" (including other terms like "contaminated site", "contaminated land", "marginal land", "polluted soil" or "polluted land") AND "minimum data set" or "MDS", and 14 hits (6 deemed relevant) for the same "brownfield" terms AND "soil quality indicator" or "SQI".

For contaminated sites, potential future land uses typically do not include crop production for agricultural purposes; however, some parts of the sites are usually transformed into green spaces for recreation which soil functions related to primary production are highly relevant (Volchko et al., 2014b). Volchko et al. (2014) highlight four proposals that suggest an MDS for evaluating soils in an urban environment with/without contamination, see Table 3-7 for the proposed indicators:

1. Schindelbeck et al. (2008) suggested an MDS for soil quality assessment for landscape management that was then applied for the soils of a vacant urban site and a more rural grass park. The MDS was purposed for 'soil health evaluation,' with emphasis on processes related to crop production.

2. Lehmann et al. (2009; 2010) suggested using different sets of SQIs for specific soil end uses, e.g., soil as (i) basis for life and habitat of flora and fauna; (ii) site for grass land use or wheat production, (iii) filter and buffer of heavy metals.

3. Bone et al. (2010) suggested an MDS of physical, chemical, and biological SQIs for prioritizing contaminated urban sites for soil remediation, which were selected based on a literature review.

4. Craul and Craul (2006) provided an MDS and practical recommendations aimed towards landscape architects and contractors for successful planting of trees in the built environment.

Table 3-7. MDSs for soil function evaluation suggested for non-agricultural use, adapted from (Volchko et al., 2014b).

(Schindelbeck et al., 2008)	(Bone et al., 2010)	(Craul and Craul, 2006)	(Lehmann, 2009; Lehmann and Stahr, 2010)
Physical soil quality indicators			
Soil texture	Soil texture	Soil texture	Soil texture
Aggregate stability (%)	Infiltration rate	Soil moisture	Depth of horizon
Available water capacity (m/m)	Presence of debris	Content of coarse fragments (%)	Available field capacity (1/m²)
Surface hardness	Soil odour	Structure of soil profile/Depth of soil layers	Content of coarse fragments (%)
Subsurface hardness (psi)	Soil colour	Slope of the surface	Structure of soil profile/Depth of soil layers
	Penetrability		Bulk density (g/cm3)
			Soil colour
			Penetration potential/rooting depth (cm)
Biological soil quality indicators			
Organic matter (%)	Organic carbon	Organic matter (%)	Organic matter (%)
Root health rating	Root presence		
Active carbon (oxidisable carbon) (ppm)	Plant cover		
Potentially mineralizable nitrogen (µgN/g dw/week)	Soil organism's presence and diversity		
Chemical soil quality indicators			
pH	pH	pH	pH
Extractable P (ppm)		Salinity (mS/cm)	Cation exchange capacity - CEC (mol/kg)
Extractable K (ppm)		Ca (ppm)	
Minor elements			

Volchko et al. (2014) also conducted a literature review to compile the SQIs used for evaluation of the effects on ecological soil functions in remediation projects, see (Volchko et al., 2013) for further description. The assessment approach of the studies evaluated combined the conventional extraction tests (i.e. measuring total concentrations in the soil) and the bioavailability tests with an assessment of SQIs related to soil functioning (Volchko et al., 2014b). The authors note that the studies largely emphasised that the goal of remediation is not only to reduce contaminants concentrations/amounts in the soil or to reduce their bioavailability and mobility, but also to restore the ecosystem functions (e.g. (Epelde et al., 2009a, 2009b, 2008)).

The work by Volchko et al. (2014) resulted in a candidate MDS for the evaluation of the effects on ecological soil functions in remediation projects, what was identified by compiling SQIs that are (i) suggested by two or more literature sources in Table 3-7, (ii) suggested by three or more literature sources found in literature review, and (iii) relatively easy to measure and interpret.

The MDS was separated into physical, biological and chemical indicators, shown below in Table 3-8.

Table 3-8. A candidate MDS for soil function evaluation in remediation projects, adapted from (Volchko et al., 2014b)

Soil quality indicators	Relevance to soil functions
Physical	**Aggregate stability of the soil**
Soil texture	Water infiltration, plant-available water and nutrient retention, aeration and root penetration. Buffering and filtering of heavy metals, the capacity of the soil to bind contaminants and thus protect from contamination
Content of coarse material	The increased content of coarse particles (>2mm) and presence of debris affect soil aggregate stability (i.e. ability to withstand falling apart when wet or hit by raindrops) as well prevent rooting, decrease plant-available water and decrease organic matter levels
Available water capacity	Water cycling. Water between the field capacity and the wilting point is the crucial factor of storing water in the soil for soil organisms between precipitations
Biological	**Biodiversity and nutrient cycling**
Organic matter content	Carbon cycling. Presence of organic matter leads to i) improvement of soil aggregate stability, water storage potential and nutrient cycling and ii) increased microbial diversity/activity and thus increased carbon sequestration
Potentially mineralizable nitrogen	Nitrogen cycling. Ability of microbial communities to supply plant-available nitrogen, a measure of biological activity
Chemical	**Nutrient retention and availability, buffering potential**
pH	The indicator revealing the level of toxicity and nutrient availability. Reflecting a potential for filtering and buffering of heavy metals
Available phosphorous	Phosphorous cycling. Macronutrient for plants and a measure of soil fertility

A soil function assessment with the suggested MDS was proposed for use to complement environmental risk assessment in remediation projects (Volchko et al., 2014b), and was expanded into a separate tool for use in site investigation – Soil Function Box (SF Box) tool (Volchko et al., 2019, 2014a). Volchko et al. (2014) argue that the value of an MDS is that it facilitates more comprehensive soil assessment in remediation projects, which should integrate the improved risk assessment and soil function evaluation in order to assure sustainable management of contaminated soil.

Gomez-Sagasti et al. (2012) state: '*due to the well-known heterogeneity and dynamic nature of the soil ecosystem, as well as the great variety of specific problems and interests associated with polluted and remediated soils, it is very hard to recommend a set of best indicators valid for all cases and situations.*' Concerning microbial properties, they advise always including measurements that provide information on the **biomass** (e.g. MBC), **activity** (e.g. respiration, potentially mineralizable nitrogen, soil enzyme activities), and **diversity** of soil microbial communities as part of a MDS (Gómez-Sagasti et al., 2012). Both biomass and activity measurements can serve as suitable independent indicators to assess soil microbial communities (e.g. effects of heavy-metal-induced stress); however, the authors maintain that 'linked' measurements such as the *metabolic quotient* (qCO_2 – ratio of CO_2 production to MBC) are frequently better indicators of heavy metal pollution than either microbial activity or biomass measurements alone. Complementing assessments with community-level physiological (CLPP, obtained with Biolog™ plates) and genetic (via PCR-DGGE) profiles as well as phospholipid fatty acid profiles (PLFA) have also proven to be useful indicators for the estimation of microbial diversity in heavy-metal-polluted and phytoremediated soils (Epelde et al., 2008; Gómez-Sagasti et al., 2012).

Another useful consideration in the selection of a minimum data set is to take into account the most frequently used bioindicators in soil monitoring programs (or those frequently used in remediation projects, as discussed in (Volchko et al., 2014b)). For example, Turbé et al. (2010) provide an extensive review of European soil monitoring programs and, while acknowledging that there is no established standard, point out the utility of the Envasso approach as a minimum set (i.e. Tier 1) of surrogate measures selected to assess the overall changes in soil biodiversity covering the three functional groups:

- **Soil ecosystem engineers:** earthworm biomass and diversity
- **Biological regulators:** springtails biomass and diversity
- **Chemical engineers:** microbial activity (respiration)

This minimum set of indicators could even be extended in some regions (i.e. including Tier 2 or 3), according to the availability of resources, to include e.g. all macro-fauna or nematodes (Turbé et al., 2010). It is also recognised that that soil biodiversity monitoring should be accompanied by measurements of soil abiotic characteristics, so as to be interpretable; including: habitat characteristics (e.g. geographical classification, land use type, climate data, groundwater level), soil properties (e.g. pH, SOC content, N content, C:N ratio, soil texture, CEC) and contamination and human-induced stress (e.g. concentration of heavy metals) (Turbé et al., 2010).

Faber et al. (2013) state that '*there is no consensus about a basic set (minimum test battery) of soil biodiversity indicators since all have useful characteristics, e.g. sensitivity, ease of use, ecological meaning, etc., that prevail under particular circumstances*.' However, some indicators are clearly used more often than others as shown in the results of the surveys they sent out to soil scientists across Europe, shown below in Figure 3-3. The authors note that few surveys were returned (135 total), indicating the general lack of structured soil biodiversity monitoring in Europe (*Note: Sweden was not included in the countries participating in the survey*). Those used most often per group include (Faber et al., 2013):

- **Fauna indicator** – nematodes
- **Microbes** – soil microbial biomass including SIR
- **Soil processes** – basal respiration including potential C-mineralization
- **Multi-endpoint tool (-omics)** – bacterial automated rRNA intergenic spacer analysis, B-ARISA.

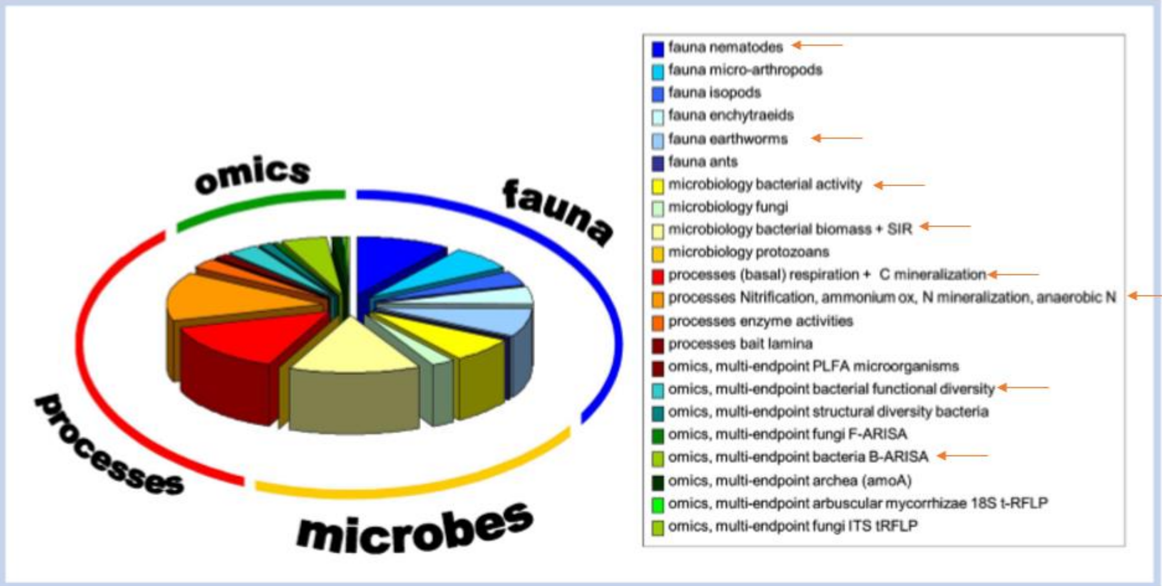


Figure 3-3. Soil biodiversity indicators used in approximately 14,200 measurements in European soils (according to results from surveys). The indicators were grouped into four categories: soil fauna (blue), microbes (yellow), soil processes (red) and 'omics' including multi-endpoint indicators like PLFA (green). Arrows (added extra) indicate those most frequently used. amoA = amoA genes in archaea (coding for the alpha-subunit of the ammonia monooxygenase); ARISA = automated rRNA intergenic spacer analysis fingerprints; ITS = internally transcribed spacer sequences; PLFA = phospholipid-derived fatty acids; SIR = substrate-induced respiration; t-RFLP = terminal restriction fragment length polymorphism. From (Faber et al., 2013).

More recently, Bünemann et al. (2018) reviewed 62 publications and extracted the most frequently proposed SQIs, see Figure 3-4. Results show that total organic matter/carbon and pH are the most frequently proposed soil quality indicators, followed by available phosphorus, various indicators of water storage and bulk density (all were mentioned in >50% of reviewed indicator sets). Soil texture, available potassium and total nitrogen are also frequently used (>40%). The authors note that the average number of proposed indicators was 11, which they consider is probably more than is feasible from a practical as well as a financial viewpoint under most circumstances (Bünemann et al., 2018). Therefore, a trend towards smaller indicator sets in recent years can be seen; however, the development of novel indicators, which can be applied on a high number of samples in a fast and cheap way, could change the picture in the future (Bünemann et al., 2018).

Bünemann et al. (2018) reported that in most of the publications at least one indicator of each category (physical, chemical and biological) is included. These categories are typically represented automatically when all soil functions or soil-based ecosystem services are addressed. However, soil biological indicators were missing from 40% of the reviewed minimum data sets. Soil physical indicators, especially those related to water storage, were frequently proposed in the early assessment schemes and again in the last 5 years, while they were less common in between. Among the soil chemical indicators, soil organic carbon content, pH, available P and K, total N, electrical conductivity, cation exchange capacity, and mineral N were proposed more often than all other indicators. Likewise, soil respiration, microbial biomass, N mineralization and earthworm density were more frequent among the biological indicators than the other 10 indicators that have been proposed at least once.

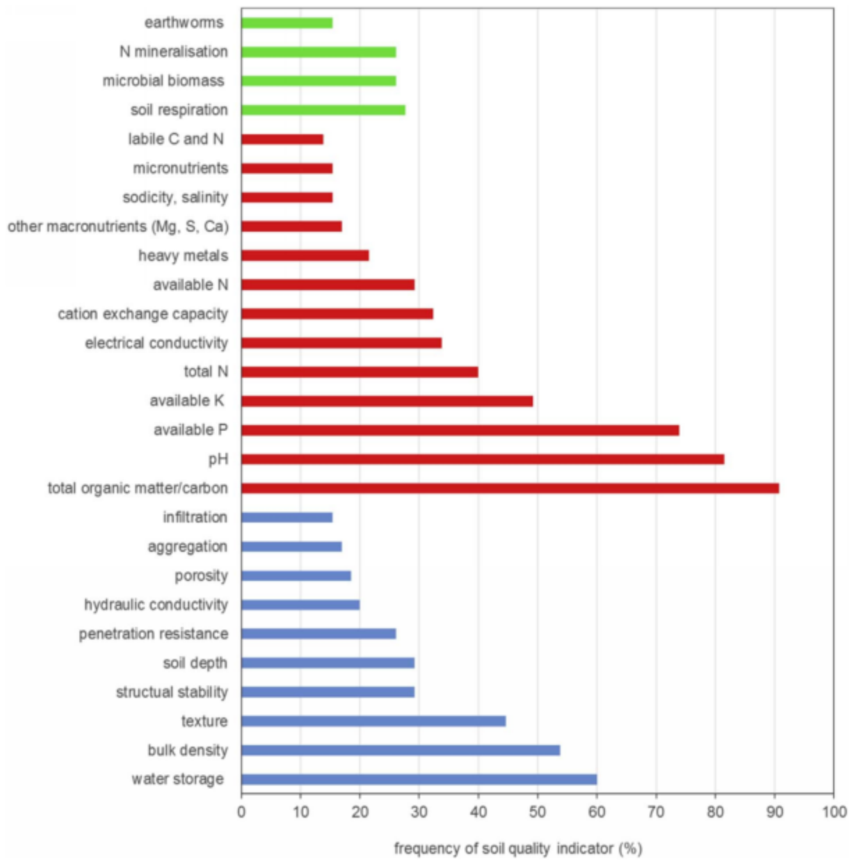


Figure 3-4. Frequency of different indicators (min. 10%) in all reviewed soil quality assessment approaches (n= 65). Soil biological, chemical and physical indicators are shown in green, red and blue respectively, from (Bünemann et al., 2018).

A crucial factor to emphasise when establishing an MDS for soil assessment is that soil organisms play a central role in soil functioning (see Appendix II); therefore, adding biological and biochemical indicators can greatly improve soil quality assessments (Barrios, 2007; Bünemann et al., 2018; Gómez-Sagasti et al., 2012). However, the more 'static', physico-chemical soil parameters ought not be neglected for, as previously stated, these parameters inform upon the condition of the soil habitat which enables life to flourish. Historically, it has been these parameters that have formed the basis for soil quality evaluations (Gómez-Sagasti et al., 2012; Ritz et al., 2009), so there is a wealth of information to draw from to inform the selection of soil quality indicators for physico-chemical conditions. In their review, Bünemann et al. (2018) note that molecular methods focusing on DNA and RNA hold great potential to perform faster, cheaper and more informative measurements of soil biota and soil processes than conventional methods. Consequently, they may yield so-called 'novel indicators' that could substitute or complement existing biological and biochemical soil quality indicators in regular monitoring programs when the performance and cost-efficiency is improved (Bünemann et al., 2018). This promise seems to be reflected in indicator selection procedures for, as previously mentioned, seven out of the top ten selected indicators in the logical sieve performed by Stone, Ritz et al. (2016) were indeed based on molecular methods, with ‘molecular bacteria and archaea diversity’ being number one. Nevertheless, soil biological indicators are still underrepresented in soil quality assessments and mostly limited to 'black-box' measurements such as microbial biomass and soil respiration (Bünemann et al., 2018). **In any case, it is**

essential to emphasize that physical, chemical, and biological properties must always be included in a proper assessment of soil quality (Gómez-Sagasti et al., 2012)**.**

3.1.4 Analyses

The key aspect to this section is **standardisation.** Römbke et al. (2018) argue that soil biodiversity indicators are critical to evaluate ecosystem functioning at a given site and to, in turn, provide ecosystem services in the face of the myriad threats that affect soil organism diversity and abundance (e.g. contamination, loss of organic matter). However, the authors also stress the importance of standardised methods in providing a robust assessment of soil quality for use in monitoring programs and other site assessments (e.g. (ISO, 2017b)), which has been echoed by many other authors (Bünemann et al., 2018; Faber et al., 2013; Griffiths et al., 2016; Turbé et al., 2010). Furthermore, since the concept of ecosystem services permeates many different fields, Römbke et al. (2018) believe that ISO should develop guidelines offering a wide range of ISO standards to be used to protect microbial functions supporting a given ecosystem service defined as specific protection goals by regulatory authorities. A recognised problem with many biological methodologies is a lack of reference control materials, which would allow for direct comparison between and within laboratories (i.e. assessment of the repeatability and reproducibility of the method – see (Creamer et al., 2009) for an inter-laboratory comparison of multi-enzyme and MSIR assays which showed significant variation) (Faber et al., 2013). Faber et al. (2013) argue that when using soil biodiversity indicators, **methods have to be standardised at least for Tier 1, preferably according to ISO standards**. The authors claim that present use of soil biological indicators in monitoring schemes is highly variable across Europe and would benefit from standardisation of sampling and identification methods (Faber et al., 2013). However, a potential *bottleneck* is the availability of labs and analyses to practically assess the parameters of interest, as evidenced by the survey questions posed in the logical sieve procedures addressing e.g. 'practicability' which refers to demands on and availability at labs (Griffiths et al., 2016; Ritz et al., 2009; Stone et al., 2016b)

Römbke et al. (2018) critically reviewed the standardised methods for assessing the structural and functional diversity of organisms, within the ISO framework, which could facilitate obtaining scientifically sound and comparable data as well as references values. A series of tables of relevant analyses were provided, shown below in Table 3-9, Table 3-10, and Table 3-11.

Table 3-9. List of ISO guidelines available to measure a range of soil physicochemical parameters, adapted from (Römbke et al., 2018).

ISO Nr	Title	Comment
10390:2005	Determination of pH	Almost all of these methods have been used in soil sciences worldwide (most in northern temperate regions) for at least 20 years by various stakeholders, including regulatory agencies, soil monitoring institutions, farmers and scientists. In parallel, they were checked every 5 years to determine whether the standard in general and/or specific parts of it still reflect the actual state of the art. In case modifications became necessary, they were improved accordingly.
10694:1995	Determination of organic and total C after dry combustion (elementary analysis)	
11260:1994	Determination of effective cation exchange capacity and base saturation level using barium chloride solution	
11261:1995	Determination of total N - modified Kjeldahl method	
11277:2009	Determination of particle size distribution in mineral soil material - method by sieving and sedimentation	
11464:2004	Pre-treatment of samples for physicochemical analysis	
11465:1993	Determination of dry matter and water content on a mass basis - Gravimetric method	
11508:1998	Determination of particle density	
13878:1998	Determination of total N content by dry combustion	
14235:1998	Determination of organic C by sulfo-chromic oxidation	
14256-1:2003	Determination of nitrate, nitrite, and ammonium in field-moist soils by extraction with potassium chloride solution. Part 1: Manual method	
14256-2:2005	Determination of nitrate, nitrite, and ammonium in field-moist soils by extraction with potassium chloride solution. Part 2: Automated method with segmented flow analysis	
15903:2002	Format for recording soil and site information	
18400-206:2017	Sampling: Guidance on the collection, handling and storage of soil for the assessment of biological, functional and structural endpoints in the laboratory	In order to update the requirements regarding the use of soil samples in ecotoxicological laboratory tests, this new standard has been prepared recently (replacing ISO 10381-6:1993)

Table 3-10. List of ISO guidelines available to estimate a range of parameters giving insight into the abundance, diversity and activity of soil fauna, adapted from (Römbke et al., 2018).

ISO Nr	Attribute	Title	Comment
23611-1:2018	Abundance and/or structure	Sampling of soil invertebrates - Part 1: Hand-sorting and formalin extraction of earthworms	This standard was recently updated.
23611-2:2006	Abundance and/or structure	Sampling of soil invertebrates - Part 2: Sampling and extraction of microarthropods (Collembola and Acarina)	This standard is still valid, however, new species identification methods ('barcoding') allowing quicker species determination must be added for all standards for individual organism groups
23611-3:2019	Abundance and/or structure	Sampling of soil invertebrates - Part 3: Sampling and extraction of enchytraeids	This standard was recently updated.
23611-4:2007	Abundance and/or structure	Sampling of soil invertebrates - Part 4: Sampling, extraction and identification of free-living stages of nematodes	Improving the practicability of this standard by modifying the sampling method is currently under discussion.
23611-5:2011	Abundance and/or structure	Sampling of soil invertebrates - Part 5: Sampling and extraction of macroinvertebrates	This standard covers a very broad range of organism groups. Thus, a new standard for surface-living species (mainly large arthropods) must be considered.
23611-6:2012	General	Sampling of soil invertebrates - Part 6: Guidance for the design of sampling programs with soil invertebrates	This standard is regularly under discussion in order to keep track with regulatory requirements.
18311:2016	Activity and/or function	Method for testing effects of soil contaminants on the feeding activity of soil-dwelling organisms - Bait-lamina test	No change necessary.
GD 56 - OECD 2006	Activity and/or function	Determination of the breakdown of organic matter in litterbags	This standard was developed for the ecotoxicological assessments of pesticides; thus, it must be modified in order to use it for the general monitoring of soil quality.

Römbke et al. (2018) observe that there is guidance for designing monitoring programs but neither sampling design nor the description of general site properties is, as of yet, standardised. However, as shown in the tables above, many of the relevant soil properties are standardised leaving the task of selecting which parameters to include in a soil assessment. An effort was recently made to implement new standards to assess the soil microbial community structure and to quantify the abundance of microbial groups from soil DNA extracts (Römbke et al., 2018). For example, different functional microbial groups (nitrifiers and denitrifiers) support the N cycle, a key component of nutrient cycling ecosystem services, have seen surging interest in monitoring and are linked to 5 standardised methodologies (i.e. ISO 14238, 15685, 17601, 20131-1, 20131-2). Of particular note, is the use of q-PCR assays to provide more sensitive functional endpoints to assess microbes involved in key functions like nutrient cycling (i.e. *functional genes*) (Thiele-Bruhn et al., 2020). Other complex molecular methods to characterize the structure and diversity of microbial communities are not yet standardized at the ISO level because most of them rely on high-throughput sequencing technologies and subsequent bioinformatics treatment of sequence data sets, which have evolved rapidly over the past 15 years (Römbke et al., 2018). According to Römbke et al. (2018), several high-throughput sequencing platforms using 'barcoding' or 'meta-barcoding' methods are on the market, but they are not stabilised yet, and there is currently no consensus among microbial ecologists about how to interpret the results. This lack of consensus inhibits their standardisation; however, the general consensus is that more of these methods should be standardised to enable widespread usage and improved assessment of functional diversity for both microorganisms and soil invertebrates (Römbke et al., 2018).

In their proposal of new microbial functional standard for soil quality assessment, Thiele-Bruhn et al. (2020) highlight the importance of microorganisms for robust soil functioning and related ecosystem services and bring attention to the fact that new standardised methods should focus more on soil microbial functions. The authors suggest 8 key soil functions and ecosystem services (based off of MEA (Millennium Ecosystem Assessment, 2005)): 1) biodiversity, genetic resources, cultural services, 2) food web support, 3) biodegradation of pollutants, 4) nutrient cycling (e.g. N, C and P), pest control and plant growth promotion, 6) carbon cycling and sequestration, 7) greenhouse gas emissions, and 8) soil structure affecting soil water, gas balance and filtration function. The few existing standardised methods available that focus on the function of soil microorganisms mostly include measurements of abundance and activity under well-defined conditions in the lab (e.g. basal respiration, nitrification, enzyme activities, biodegradation of organic matter) (Römbke et al., 2018; Thiele-Bruhn et al., 2020). For invertebrates, the only available functional tests related to the activities of these animals are: the bait-lamina (ISO, 2018) and the litter bag test (GD 56, OECD 2006) (Römbke et al., 2018). Thiele-Bruhn et al. (2020) assessed existing ISO standards for determining potential microbial biomass and activities and connect to the functions listed above, shown in Table 3-11:

Table 3-11. List of ISO methods available to estimate a range of parameters, giving insight into the abundance, structure and activity of soil microorganisms, adapted from (Römbke et al., 2018) *and adding functional relevance from* (Thiele-Bruhn et al., 2020).

ISO Nr	Attribute	Title	Comment	Functional relevance
10832:2009	Activity	Effects of pollutants on mycorrhizal fungi - spore germination test	This standard was last reviewed and confirmed in 2016.	N/A
11063:2012	Required for abundance and diversity analyses	Method to directly extract DNA from soil samples	This standard was last reviewed and confirmed in 2017.	Biodiversity, genetic resources; Carbon cycling and sequestration
11266:1994	Activity	Guidance on laboratory testing for biodegradation of organic chemicals in soil under aerobic conditions	This standard was last reviewed and confirmed in 2016.	Biodegradation of pollutants
14238:1997	Activity	Determination of N mineralisation and nitrification in soils and the influence of chemicals on these processes	This standard was last reviewed and confirmed in 2017.	Nutrient cycling
14239:2017	Activity	Laboratory incubation systems for measuring the mineralisation of organic chemicals in soils under aerobic conditions	This standard was last reviewed and confirmed in 2017.	Biodegradation of pollutants
14240-1:1997	Abundance	Determination of soil microbial biomass - Part 1: substrate-induced respiration method	These 3 standards were last reviewed and confirmed in 2014.	Carbon cycling and sequestration
14240-2:1997	Abundance	Determination of soil microbial biomass - Part 2: fumigation-extraction method		Carbon cycling and sequestration
15473:2002	Activity	Guidance on laboratory testing for biodegradation of organic chemicals in soil under anaerobic conditions		Biodegradation of pollutants

15685:2004	Activity	Determination of potential nitrification and inhibition of nitrification - rapid test by ammonium oxidation	This standard was last reviewed and confirmed in 2017.	Nutrient cycling
16072:2002	Activity	Laboratory methods for determination of microbial soil respiration	This standard was last reviewed and confirmed in 2014.	Carbon cycling and sequestration
17155:2012	Activity	Determination of abundance and activity of soil microflora using respiration curves	This standard was last reviewed and confirmed in 2017.	Carbon cycling and sequestration
17601:2016	Abundance	Method to quantify the abundance of microbial communities from soil DNA extracts	This standard was published in 2016 and remains current.	Biodiversity, genetic resources; Carbon cycling and sequestration
20130:2018	Activity	Measurements of enzyme activity patterns in soil samples using colorimetric substrates in micro-well plates	Under development at time of writing - now complete	Carbon cycling and sequestration; Nutrient cycling
20131-1:2018	Activity	Assessment of the capacity of soils to reduce N2O - Part 1: soil denitrifying enzyme activities		Greenhouse gas emissions
20131-2:2018	Activity	Assessment of the capacity of soils to reduce N2O - Part 2		Greenhouse gas emissions
22939:2019	Activity	Measurement of enzyme activity patterns in soil samples using fluorogenic substrates in micro-well plates		Carbon cycling and sequestration; Nutrient cycling
23753-1:2019	Activity	Determination of dehydrogenase activity in soils - Part 1: Method using triphenyltetrazolium chloride (TTC)	These standards were last reviewed and confirmed in 2019.	Carbon cycling and sequestration; Nutrient cycling
23753-2:2019	Activity	Determination of dehydrogenase activity in soils - Part 2: Method using iodotetrazolium chloride (ITC)		Carbon cycling and sequestration; Nutrient cycling
29843-1:2010	Structure	Determination of soil microbial diversity - Part 1: Method by PLFA analysis and PLEL analysis	This standard was last reviewed and confirmed in 2017.	Biodiversity, genetic resources; Carbon cycling and sequestration
29843-2:2010	Structure	Determination of soil microbial diversity - Part 2: Method by PLFA analysis using the 'simple PLFA extraction method'	This standard was last reviewed and confirmed in 2015.	Biodiversity, genetic resources; Carbon cycling and sequestration

Methods to assess the biodegradation of pollutants are already implemented into ISO guidelines (e.g. ISO 14239, 15473 – assessing the degradation of organic chemicals), and are part of legal frameworks including pesticide directives (Thiele-Bruhn et al., 2020). In previous years, the development of standard methods was mainly driven by the need to assess the ecotoxicological effects of anthropogenic activities, such as chemical contamination of soils, rather than to describe and understand the natural properties and functions of soils (Thiele-Bruhn et al., 2020). Defining methods for the determination of adverse effects of contaminants on soil biota was also a major task of other organisations such as the Organization for Economic Co-Operation and Development (OECD)[8]. For example, there are OECD guidelines (OECD 216 and 217

[8] Note: According to Römbke et al. (2018): Standardisation in the field of soil biodiversity is mainly carried out by ISO. Other international standardization institutions (e.g., the Organisation for Economic Co-operation and Development [OECD]) focus on ecotoxicological tests to assess chemicals, but they are far less active in soil biodiversity. OECD and ecotoxicological tests were not discussed in their review though both are still relevant to site assessment.

(OECD, 2000a, 2000b)**)** for testing the long-term effects of single-exposure chemicals on soil microbial nitrogen and carbon transformation, respectively.

3.1.5 Interpretation of indicator values

As stated by Bünemann et al. (2018), '*an indicator is only useful if its value can be unequivocally interpreted and reference values are available*.' Reference values for a given indicator could be either those of a native soil, which may however not be suitable for agricultural production, or of a soil with maximum production and/or environmental performance. Acceptable values for an indicator can also be defined as those at which there is no loss or significant impairment of functioning; in the context of pollution, thresholds of contamination are often used (Bünemann et al., 2018).

A more advanced way to evaluate soil quality indicators is the establishment of standard non-linear scoring functions, which typically have the shapes i) more is better, ii) optimum range, iii) less is better, or iv) undesirable range (as shown in Figure 3-6), with i-iii being most common in soil science (Bünemann et al., 2018). The shape of such curves is established based on a combination of literature values and expert judgement (Andrews et al., 2004). When scoring curves are based on regional data, such as in the Cornell Soil Health Assessment (Moebius-Clune et al., 2016), then scores are relative to measured values in the respective region. Each indicator measurement is transformed to a value between 0 and 1 (or 0 and 100) using a scoring algorithm, with a score of 0 being the poorest (lower threshold) and a score of 1 (or 100) the best (upper threshold). The baseline value equals the midpoint between threshold values. Validation of scoring curves is possible if datasets with measurements of the given soil quality indicator and a related soil process are available (Andrews et al., 2004; Bünemann et al., 2018; Moebius-Clune et al., 2016; Volchko et al., 2014a).

Bünemann et al. (2018) maintain that acceptable target ranges of soil quality indicators need to be soil-and land use-specific, and they depend not only on targeted soil functions, but also on both spatial and temporal scale of soil quality assessments, with regional target ranges typically being narrower than national ones. In addition, acceptable ranges of a soil quality indicator for one property or process are often highly dependent on the value of another soil property or process, e.g. dependence of microbial biomass or soil organic carbon on soil texture. Due to the contentious nature of interpreting SQIs (e.g. establishing target values, data limitations and difficulties interpreting values that often rely on expert judgement), a comparative approach in which indicator values or scores of a given sampling point are put in relation to other sampling points may be the most intuitive and flexible basis for interpretation (Bünemann et al., 2018). Providing a relative assessment (e.g. top 25%) in this way allows continuing evolution of the system and ease of understanding (Bünemann et al., 2018). For example, in the *Applicera* project, contaminant levels (toxic pressure, indicating toxicity in relation to threshold values) were plotted against SF Box scores of potential soil quality (soil classes representing good, medium and poor quality) (Volchko et al., 2020), see Figure 3-5.

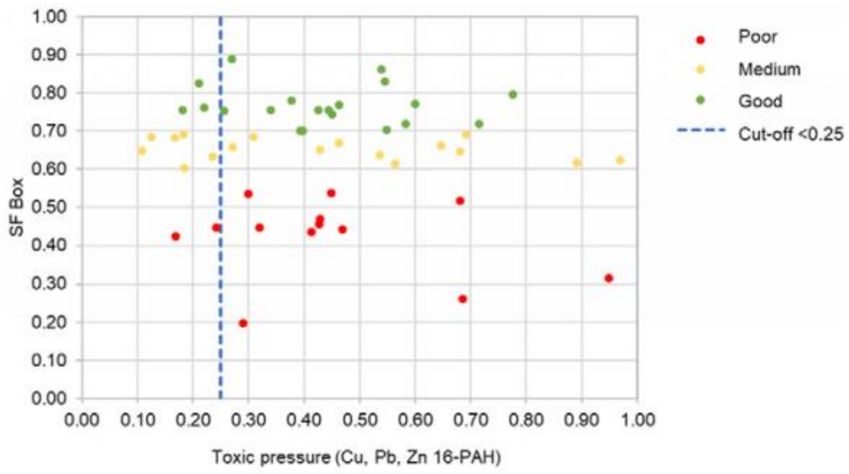


Figure 3-5. Soil quality obtained with SF Box as a function of bioavailability corrected toxic pressure (TP) values. Samples were grouped into three soil quality classes according to the SF Box index. Sampling points with TP < 0.25 were used as reference points in the ecological line of evidence, from (Volchko et al., 2020).

Similar to the 'SF Box score' shown above, many studies on soil quality have searched for a way to aggregate the information obtained for each soil quality indicator into a single soil quality index (Andrews et al., 2004; Bünemann et al., 2018; Epelde et al., 2014b; Velasquez et al., 2007). The ultimate purpose of a soil quality index is to inform farmers and other land managers about the effect of soil management on soil functionality, which is an especially useful communication tool, particularly by graphical means, as an aggregated presentation of the outcome of soil quality assessments communicating that can transmit to society as a whole the consequences that human decisions can have on soil-based ecosystem services (Bünemann et al., 2018). According to Turbé et al. (2010), no comprehensive index has yet been proposed that would combine all the aspects of soil complexity into a single formula and allow accurate comparison among sites and plots. Existing indicators comprise rather long lists of potentially relevant variables to be measured, although no general agreement has been reached on their interpretation (Doran and Zeiss, 2000; Turbé et al., 2010). Some attempts to combine groups of variables into indicators of soil biotic activity have recently been proposed, which can be classed into three main approaches (see (Turbé et al., 2010) for more detail and compilation of compound indicator systems):

- **Shopping list approach** – where a set of different soil parameters are assessed using 'simple indicators'
- **Benchmark approach** – where the degree of deviation between reference situations and the actual measurements are evaluated using compound indicators (*often limited by absence of reference systems and indicators)*
- **Numerical approach** – where synthetic indices are developed for the assessment of soil status using compound indicators

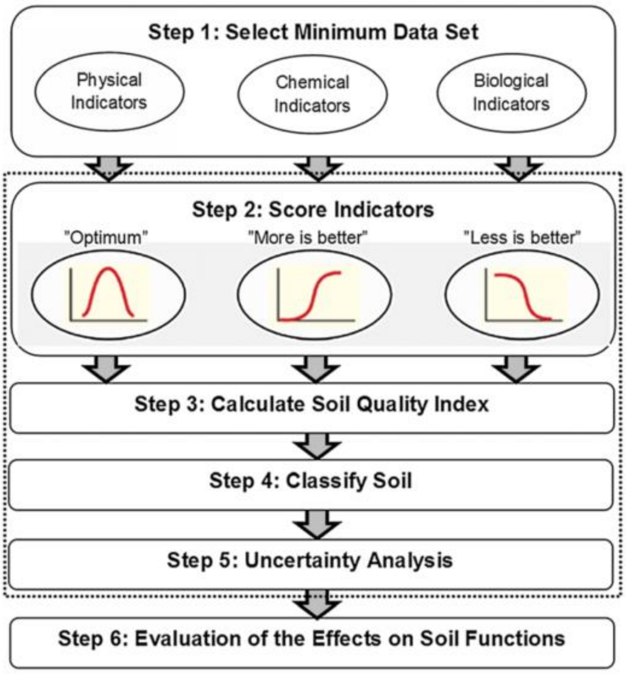


Figure 3-6. A generic framework for soil function assessment. The dotted box corresponds to operations carried out in the SF Box tool, from (Volchko et al., 2014a).

4 Ecosystem services assessment

There has long been disagreement over defining exactly what ecosystem services (ES) are, but they can generally defined as *the direct and indirect contributions of ecosystems to human well-being* (Potschin and Haines-Young, 2016; TEEB, 2010). The ES concept is becoming mainstream in policy and planning for communicating about the environment and operationalisation has even led to gradual changes in decision-making and action (Dick et al., 2018). Many international research initiatives and classification systems such as The Millennium Ecosystem Assessment (MEA) (Millennium Ecosystem Assessment, 2005), The Economics of Ecosystems and Biodiversity (TEEB) (TEEB, 2010), The Common International Classification of Ecosystem Services (CICES) (Haines-Young and Potschin, 2018; Potschin and Haines-Young, 2016) and Mapping and Assessment of Ecosystems and their Services (MAES) (Maes et al., 2016) emphasise that the concept of ES is fundamentally a reductionist, anthropogenic view on nature and acknowledges the instrumental value of ecosystems (i.e. its components, processes and functions) as a means of sustaining and improving human well-being. A true, holistic accounting of the value of the ecological system must account for the non-anthropocentric or *intrinsic* value of ecosystems regardless of the benefits it delivers to humans (Back et al., 2019; La Notte et al., 2017). In the context of soils, their intrinsic value (e.g. as a haven for biodiversity, existence value) has been argued from both an ethical and practical perspective by Back et al. (2019) as worthy to be included under the umbrella of protection offered by such agendas as the Soil Security framework, which is predominantly focused upon human well-being (Back et al., 2019). The authors highlight three cases where the non-anthropocentric value of soil is especially important (in the context of contaminated soils): 1) when ES are difficult to assess or are of low value, 2) when the intrinsic value of soil is especially strong, and 3) when human activity has caused negative effects on soil organisms or on other parts of the ecosystem (e.g. secondary poisoning in the food web). They argue ignoring the non-anthropocentric value of soils in these situations could actually counteract sustainability efforts and even allow threats to persist (Back et al., 2019).

There are many standardised classification systems but no one that has been universally agreed upon. MEA, TEEB and CICES are the most prominent in the field though even these have slightly different classifications of the individual ecosystem services, the scale of grouping and what is considered an ecosystem service in various contexts (e.g. soil ES, urban, ES, etc.). MEA and TEEB provide much broader, simplified classification systems while CICES breaks down the broader categories progressively into smaller groupings that can be used in economic and environmental accounting to facilitate assessment and valuation (Haines-Young and Potschin, 2018; Potschin and Haines-Young, 2016). The Swedish Environmental Protection Agency (SEPA) based their ES classification system on the CICES framework (SEPA, 2017). Table 4-1 provides a comparison between the three main classification systems:

Table 4-1. Ecosystem services classification systems – comparing MEA, TEEB and CICES.

Category	MEA	TEEB	CICES (v 5.1)
Provisioning	Food	Food	*Biomass* – Cultivated or wild terrestrial and aquatic plants (incl. algae, fungi) and animals for nutritional purposes
	Fibre, Timber, Ornamental, Biochemicals	Raw materials, Medicinal resources	*Biomass* – Cultivated or wild terrestrial and aquatic plants (incl. algae, fungi) and animals for direct use or processing as raw materials or energy
	Fresh water	Fresh water	*Water* – Surface or groundwater used for nutrition, materials or energy

	Genetic resources	Genetic resources	*Genetic materials from all biota* – Plants, algae, fungi, animals or other organisms
Regulating & Maintenance	Water purification and waste treatment, Air quality maintenance	Wastewater treatment (water purification), Air quality regulation	*Transformation of biochemical or physical inputs to ecosystems* – Mediation of wastes or toxic substances of anthropogenic origin by living processes
			Transformation of biochemical or physical inputs to ecosystems – Mediation of nuisances of anthropogenic origin (e.g. smell, noise, aesthetic)
	Water regulation, Storm protection, Erosion control	Regulation of water flows, Moderation of extreme events, Erosion prevention	*Regulation of physical, chemical, or biological conditions* – Regulation of baseline flows and extreme events (e.g. erosion, water regulation, storm protection, wind protection, fire protection)
	Pollination	Pollination	*Regulation of physical, chemical, or biological conditions* – Lifecycle maintenance, habitat and gene pool protection
	Biological control, Regulation of human diseases	Biological control	*Regulation of physical, chemical, or biological conditions* – Pest and disease control
	Soil formation and retention (supporting)	Maintenance of soil fertility	*Regulation of physical, chemical, or biological conditions* – Regulation of soil quality
	Water cycling (supporting)		*Regulation of physical, chemical, or biological conditions* – Water conditions
	Climate regulation, Air quality maintenance	Climate regulation, carbon sequestration and storage, Air quality regulation	*Regulation of physical, chemical, or biological conditions* – Atmospheric composition and conditions
			Other types of regulation and maintenance services by living processes
Cultural	Recreation and ecotourism	Tourism, Recreation and mental and physical health	*Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting* – Physical and experiential interaction with the natural environment
	Aesthetic values, Inspiration, Cultural diversity, Cultural heritage, Social relations, Knowledge systems, educational values	Aesthetic appreciation and inspiration for culture, art and design, Recreation and mental and physical health	*Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting* – Intellectual and representative interactions with the natural environment (e.g. traditional knowledge, culture, education, aesthetics)
	Spiritual and religious values, Sense of place	Spiritual experience and sense of place	*Direct, in-situ and outdoor interactions with living systems that do (or do not) depend on presence in the environmental setting* – Spiritual, symbolic and other interactions with the natural environment
			Direct, in-situ and outdoor interactions with living systems that do not require presence in the environmental setting – Other biotic characteristics that have a non-use value (e.g. existence, bequest)
Supporting	Primary production		*Regulation of physical, chemical, or biological conditions* – Lifecycle maintenance, habitat and gene pool protection

	Nutrient cycling		*Regulation of physical, chemical, or biological conditions* – Regulation of soil quality (soil formation and composition)
	Habitat provisioning	Habitats for species, Maintenance of genetic diversity	*Regulation of physical, chemical, or biological conditions* – Lifecycle maintenance, habitat and gene pool protection
Purpose:	*MEA provides a classification that is globally recognised and used in sub global assessments.*	*TEEB provides an updated classification, based on the MEA, for urban ecosystem services.*	*CICES provides a hierarchical system, building on the MA and TEEB classifications but tailored to accounting. The ES listed correspond to Division - Group in CICES designation which can be further broken down into specific classes.*

When considering the benefits that ES provide humans, there is a consensus that there exists some kind of 'pathway' for deriving human well-being from ES that are delivered by ecological structures and processes (Potschin and Haines-Young, 2016). The framework for assessing the delivery of ES to humans is often portrayed as a 'cascade' model that captures that view of a 'production chain' linking biophysical structures and processes to ecosystem functions ecosystem services, benefits and values that a considered system provides to humans (Andersson-Sköld et al., 2018; Potschin and Haines-Young, 2016, 2011; TEEB, 2010). As shown in Figure 4-1, for the example system of an urban woodland or park, clear connections can be drawn between the functions that the woodland has the capacity to provide (e.g. slowing the passage of water) and the service this provides humans (e.g. reducing flooding in cities) thereby providing benefits which can be valuated (Andersson-Sköld et al., 2018; Potschin and Haines-Young, 2016). The various ES classification systems (i.e. CICES) aim to classify so-called 'final' ecosystem services that link to the goods and benefits that are valued by people and contribute to human well-being (Potschin and Haines-Young, 2016). CICES considers supporting services or ecological functions separately as the underpinning structures and processes that ultimately give rise to ecosystem services; instead, CICES provides a classification of potential final services which can be attributed to *direct* human benefit and valued appropriately (Potschin and Haines-Young, 2016).

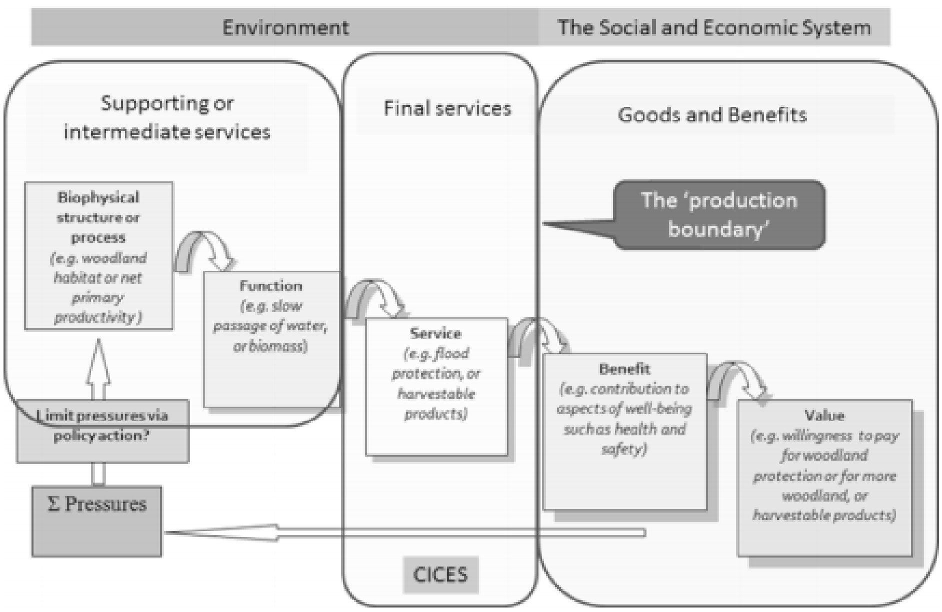


Figure 4-1. The cascade model framework for ecosystem valuation, from (Potschin and Haines-Young, 2016).

This model is not without criticism, and practical inconsistencies and confusion regarding ES frameworks, classification systems and terminology abound which can hinder consistent application (La Notte et al., 2017).

Regarding the assessment of soil functions and soil-based ES, reviews by Baveye et al. (2016) and Greiner et al. (2017) assess the state of the field and methods for quantifying and valuing them. They conclude that, thus far, soil has not been well represented in ES assessments (Baveye et al., 2016; Greiner et al., 2017); even though it has been acknowledged that soil is an integral to ES provisioning, especially considering that 'soil fertility' and 'soil formation' are commonly mentioned in ES frameworks (Dominati et al., 2010; Greiner et al., 2017). Soil functions are typically included in soil quality assessments (including closely related concepts such as soil health, soil protection, soil security, etc.) so there is a degree of overlap in these related fields (Greiner et al., 2017). As previously noted, the capacity of soils to deliver ES is largely determined by its functions; however, operational tools for quantifying soil-related ES in ES assessments is still lacking (Greiner et al., 2017). To demonstrate the contributions soils make to ES, Greiner et al. (2017) used a similar cascade model as that shown above to develop linkages between soil functions and ES, see Figure 4-2. Their model emphasises the importance of soil properties and processes to the delivery of ES, many of which can be used to assess individual soil functions (Greiner et al., 2017). In their review, they identified eight soil-based ES which were most frequently included in ES mapping studies (in order of frequency): 1) carbon pool, 2) agricultural production, 3) water cycling, 4) nutrient cycling, 5) habitat for plants, 6) filter/buffer of organic compounds, 7) filter/buffer of inorganic compounds, 8) acidity buffer. Many studies neglected the inherent multi-functionality of soils to deliver many ES (with notable exceptions), and instead narrowed in on a few of the most commonly considered soil functions (Greiner et al., 2017).

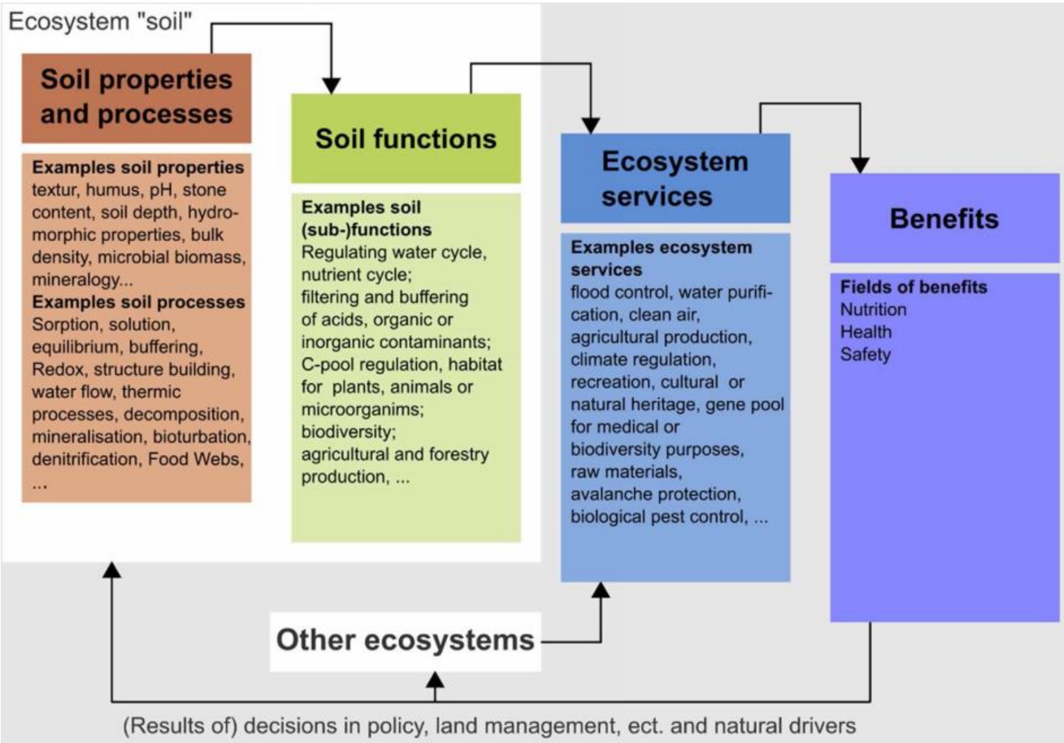


Figure 4-2. Assessment of the contributions of soil functions to ecosystem services using the cascading framework, from (Greiner et al., 2017).

4.1 Assessment methods

In the context of contaminated soils, application of ecosystem service assessment or valuation methods is still limited to only a few studies (De Valck et al., 2019). Those which have been considered to be most instructive will be briefly discussed in this section. These methodologies can be separated into essentially two types of procedures:

1. **Quantitative analysis –** based upon the use of assessment endpoints and indicators linked directly to specific soil functions and ecosystem services.
2. **Ecosystem service mapping** – procedure utilising a semi-quantitative, quantitative or qualitative approach based upon using proxies or stand-in indicators for estimating the expected change in ecosystem services at a site that could result from some land management alternative, e.g. a remediation option.

4.1.1 Quantitative analysis

In this section, analytical methods for quantitatively assessing ES have been separated into those based upon either 1) ecological risk assessment or 2) integrative numerical indices.

Ecological risk assessment

Traditionally, ecological risk assessment (ERA) focuses on assessing the risk of chemical contamination by comparing soil contaminant (pseudo-)total concentrations to soil quality standards derived from laboratory-based ecotoxicity experiments based on 'species sensitivity distributions' (SSD), which estimate the 'potentially affected fraction' (PAF) of a standard set of species from a chemical stressor (Faber and Van Wensem, 2012; Gutiérrez et al., 2015; Volchko et al., 2014b, 2014a). However, such an approach, named ‘Tier 1’ ERA, may not provide accurate, unbiased results given that field conditions are likely to differ from the laboratory conditions used to derive soil-quality standards (i.e. total concentrations versus bioavailable concentrations), most toxicological data used in SSDs is for a select few taxonomic groups, ecological complexity is neglected and that various confounding factors can affect actual contaminant ecotoxicity within soil (e.g. mixtures of contaminants) (Faber and Van Wensem, 2012; Gutiérrez et al., 2015; Volchko et al., 2020, 2014a). Biodiversity has so far been neglected in most environmental assessments because it is often considered too broad and vague a concept to be applied to real-world regulator and management problems (Turbé et al., 2010). Additionally, many soil investigations and remediation projects do not consider soil properties unrelated to contamination (such as nutrient or water availability) and may therefore result in undesirable economic and environmental outcomes (Volchko et al., 2020).

It has been recognised that the sole use of SSDs is not sufficient to assess ecological risks, thus encouraging the use of expanded, site-specific ERAs like the Triad approach (e.g. (ISO, 2017b; Mesman et al., 2006)) (Faber and Van Wensem, 2012; Gutiérrez et al., 2015); where different 'lines of evidence' (e.g. chemical, biological, toxicological analyses) stand on their own and provide individual pieces of the much larger puzzle that comprises the natural environment. Concern has also been expressed about the lack of impact of ERA on decision-making as well as the fact that decision makers have not been appropriately focused on ecological endpoints (Faber and Van Wensem, 2012). Over the past decade, developments in environmental monitoring and risk assessment to remedy this situation converged towards the use of indicators and **endpoints** (defined as *an explicit expression of the environmental value to be protected, operationally defined as an ecological entity and its attributes* (US EPA, 2016)) that are related to soil functioning and ecosystem services (Faber et al., 2013). Chapman (2008) argues that ecosystem services are effectively *assessment endpoints* (i.e. the part of the natural environment that we are trying to assess and protect). While it may not be possible to measure them directly, we can measure certain components of the ecosystem (i.e. *measurement endpoints*) to give us a reasonable indication of the health of the whole (Chapman, 2008). Thomsen et al. (2012) argue that ERA is needed to facilitate the assessment of soil health and the capability of a soil to provide ecosystem services for a desired biologically active end use (i.e. 'suitability for use') such as e.g. detoxification and decomposition of wastes, soil formation and renewal of soil fertility. An example, from Semenzin et al. (2009), of how the taxonomic group-ecological process relationships might be incorporated into Triad LoE is shown in Figure 4-3.

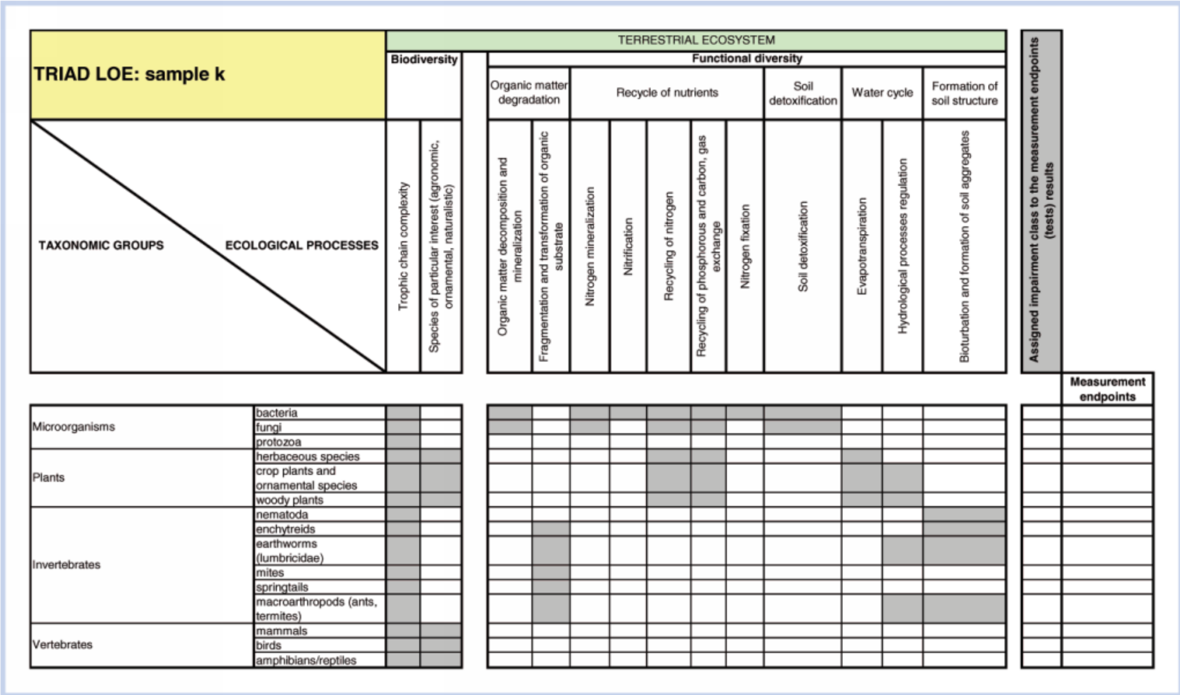


Figure 4-3. Ecosystem Impairment Matrix (EcoIM), as implemented in DSS-ERAMANIA Module 2, that shows the integration of taxonomic group-ecological process relationships into Triad LoE, from (Semenzin et al., 2009a).

One of the most promising avenues for the integration of ES within ERA has been advocated by Faber et al. (2012; 2013), proposing to use assessment endpoints and indicators that are clearly linked to land use objectives (i.e. stakeholder demands) and representative for specific ES thereby enhancing the societal relevance of the ERA and its value in decision-support. They explain that the goal is not necessarily to change the way assessment studies are usually conducted, but rather to **shift the focal point of assessment away from biodiversity and ecosystem health towards ecosystem services and associated benefits relevant for society** (Faber and Van Wensem, 2012). According to Forbes and Calow (2013), the strength in formulating protection goals in ERA in terms of ES is in contributing to management by connecting ecosystem structure and processes to what is valued (i.e. human well-being), thus making risk assessment more policy- and value-relevant. It is argued that deriving assessment endpoints from structures and processes in the ecosystem that are considered indispensable for the provision of particular provides several distinct advantages, including i) *communicative strength* – to better explain the value of ecosystems for mankind, becoming more policy relevant and enabling different actors to speak a common language, ii) *overarching/integrative aspects* – using similar data and assessment processes across fields, e.g. equating ecological risk with ES loss or including ES in ERA (Munns et al., 2009), including developing SQI for use in ERA to create broader soil management policy, iii) *possibility to value ES* – in monetary terms or otherwise, is another aspect that makes the concept attractive for decision makers as it may facilitate the weighing of options (e.g. benefits of risk reduction measures or restoration of contaminated soils for society via cost-benefit analysis e.g. (Volchko et al., 2020)) (Chapman, 2008; Faber et al., 2013; Faber and Van Wensem, 2012).

Faber et al. (2012; 2013) maintain that the concept of ES can be used as a guiding principle in environmental quality assessment, see Figure 4-4. A crux is the identification of relevant indicators of ecosystem services, through recognition of essential structures and processes that are key in the delivery of these services (Faber et al., 2013). Harmonisation in approaches utilising ES will benefit communication about assessments using biological indicators (Faber et al., 2013; Thomsen et al., 2012). The applications of ES shown in Figure 4-4 refer to the following scenarios:

A) Literature toxicity data for relevant indicators may be used for derivation of soil quality criteria for sustainable land use – scenarios may be selected for different types of land use where related ES and associated indicators are selected. In the derivation process for chemical soil quality objectives, those indicators will be selected that are sensitive and have been used in toxicity testing. The toxicity data may be compiled from literature to make up datasets that can be used in the traditional way of deriving soil quality objectives (e.g. (Thomsen et al., 2012))

B) Site-specific ERA may use the status of ES indicators to assess suitability for use of the soil. Given the desired type of land use and specific aims of local stakeholders, vulnerable indicators are selected to be used in bioassays or to be assessed through field inventory and monitoring in a Triad approach. Notably, indicators should be selected as relevant as can be in view of the intended land use and management practices in question (e.g. relevant for the type of agricultural use in terms of cropping and soil management).

C) Indicators for soil biological monitoring may be selected with respect to relevant ecosystem services required in associated to land use. Biological measures can be used as indicators of impact and change both to biodiversity itself and associated ES pertinent to specific types of land uses.

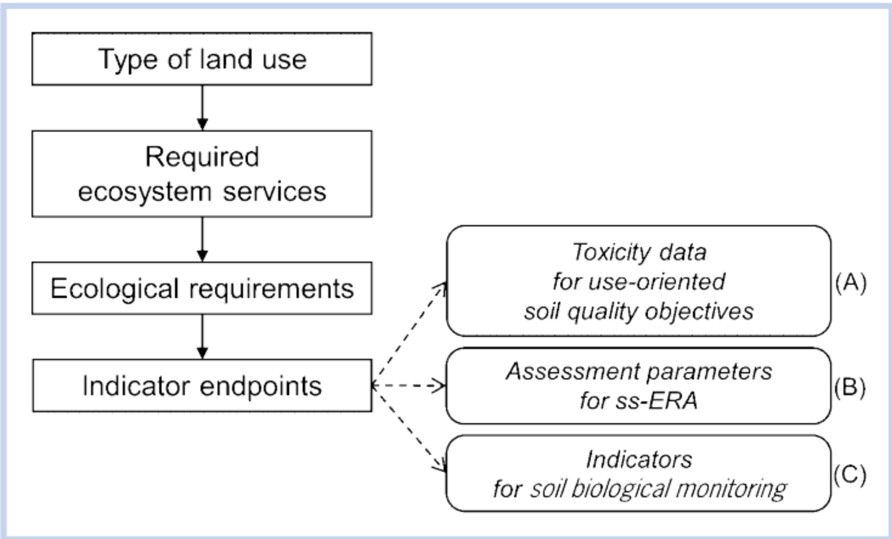


Figure 4-4. Derivation of relevant indicators for evaluation of ecosystem service provision in environmental quality assessment, through ecological requirements that are conditional for sustainable land use of specific type, from (Faber et al., 2013).

There seems to be a general consensus that **when a land (or soil quality) management strategy incorporates the concept of ecosystem services, quantifiable soil features can be more easily linked to land-use expectations and protection goals in a defensible and transparent way** (Bünemann et al., 2018; Burges et al., 2018, 2016; Epelde et al., 2014a, 2014b; Faber et al., 2013; Faber and Van Wensem, 2012; Garbisu et al., 2011; Gómez-Sagasti et al., 2012; Gutiérrez et al., 2015; Pulleman et al., 2012; Rutgers et al., 2012). Kapustka et al. (2016) argue that assessment endpoints should be related to ecosystem services because they provide a common currency for determining what priority ecological attributes to protect within ecosystems and across management goals. For example, measurement endpoints used to characterize the existing ecological conditions for selected ecosystem services can also be used to evaluate restoration (or remediation) success (Kapustka et al., 2016; Rohr et al., 2016). This also relates to the problem posed by Faber et al. (2012), stating that, while soil ecosystems are infinitely complex, the question (in risk assessment) is essentially *What to protect?* This question can be a prime cause for miscommunication between land using stakeholders and scientific risk assessors as well as lead to a lack of acceptance of results of ERA (Faber and Van Wensem, 2012). In determining the 'level of protection' at a site, the more specific question to answer is *Which factors determine or modulate the risk to the soil ecosystem from pollution and other stressors?* (Faber and Van Wensem, 2012).

Regarding investigation at contaminated sites, the integration of ES into ERA (scenario B) has been discussed by many authors (e.g. (Faber, 2006; Faber et al., 2013; Faber and Van Wensem, 2012; Forbes and Calow, 2013; Galic et al., 2012; Gutiérrez et al., 2015; Kapustka et al., 2016; Munns et al., 2009, 2016; Niemeyer et al., 2012; Semenzin et al., 2009a; Slack, 2010; Swartjes et al., 2011; Thomsen et al., 2012; Volchko et al., 2020)). A search in the Scopus database for "ecological risk assessment" AND "ecosystem services" resulted in 87 hits with 14 of these deemed relevant after reading through abstracts. While all the relevant articles are instructive, two of the proposed methods were particularly relevant to this study and will be discussed in more detail:

Thomsen et al. (2012) – Problem tree conceptual model linking:

Land use → ecosystem services → ecological requirements → indicators

Thomsen et al. (2012) provide an approach (in answer to the above-mentioned questions) that was aimed to tailor risk assessment in alignment with intended land use and concurring demands for soil quality. The assessment focuses on characteristics and vulnerability of the biological community required for soil ecosystem functioning (Faber and Van Wensem, 2012; Thomsen et al., 2012). The method is based upon a conceptual system model, a 'problem tree configuration' – see Figure 4-5, for describing soil ecosystem health in terms of 'suitability for use' (i.e. the provision of ecosystem services), using defined ES as proxies for soil health (i.e. assessment endpoints). It utilises ecotoxicological data to systematically identify specific, 'vulnerable' (i.e. sensitive) indicators that are associated with essential ecosystem structures and processes that underlay soil ecosystem services (i.e. 'ecosystem service providers'). The authors note that impacts of chemical stress (i.e. contamination) on biological diversity and land use impacts on soil fertility or health are complex and difficult to quantify; nonetheless, such impacts are drivers for decreased habitat quality to a level that may exceed soil ecosystems ability to maintain the ecological processes and functions mediating e.g. waste assimilation and detoxification and primary productivity (Thomsen et al., 2012). The key assumption made is that if indicators reflecting ecological requirements for ES are not exposed beyond their no-effect concentration thresholds, the associated ES is not impacted (Thomsen et al., 2012).

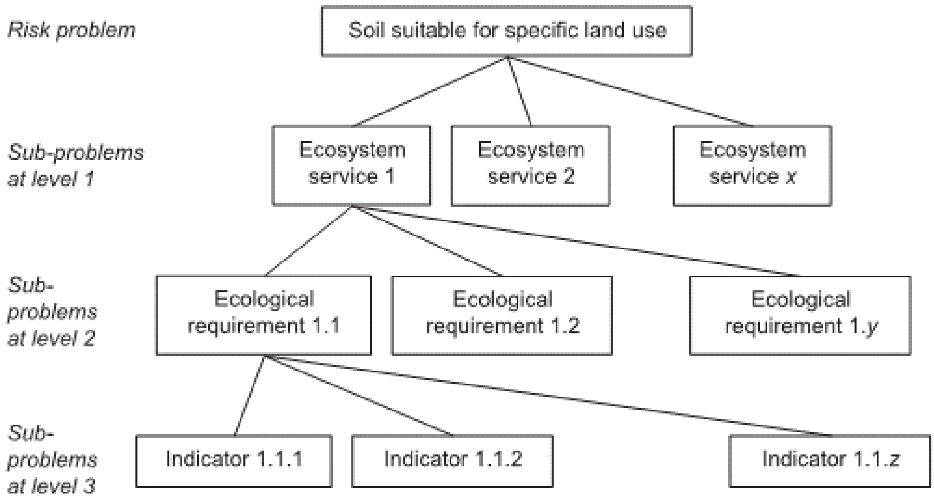


Figure 4-5. Problem tree configuration for soil ecosystem health in terms of ecological suitability for specific land use, depicted as ecosystem services subdivided into ecological requirements down into ecological indicators representing ecosystem structure and functioning, biodiversity and soil processes that are needed for the provision of soil ecosystem services, from (Thomsen et al., 2012)

The problem tree is broken down into the following stages:

- **Risk problem: suitability for land use** – risk assessment of contaminated land may focus on ecological suitability for land uses (e.g. agricultural, nature (meadow), and urban (recreational greenspace)) entailing provisioning of select ES. This stage will gain in accuracy by precise definition of intended land use and goals for use, thus enabling selection of suitable ES.

- **Sub-problems level 1: ecosystem services** – soil ecosystem health is not easily expressed in well-defined representative measurable sub-problems (i.e. biological, physical and chemical key parameters that need to be protected for the soil system to survive. Also, due to of the complexity of the soil ecosystem, ecological indicators might not be available at all; however, the degree of fulfilment of ES can be used as proxies for ecosystem health. The following 6 ES were identified by Thomsen et al. (2012) as relevant for all open types of land use:

1. *Soil fertility* – the capacity to provide nutrients and biomass
2. *Adaptability and resilience* – the capacity to adapt, or the fragility upon disturbance, and changes in land use
3. *Buffer and reaction function* – storage and buffering of water, gases, chemicals, energy, CEC, breakdown and synthesis of chemicals (detoxification, humification)
4. *Biodiversity and habitat provision* – genetic, functional and structural diversity
5. *Disease suppression and pest resistance* – the natural capacity to prevent and suppress pests and diseases
6. *Physical structure* – supportive capacity, historical archive, landscape identity

- **Sub-problems level 2: ecological requirements** – for ES to be provided to society, a certain level of soil quality is required. In this framework, 'ecological requirements' are the actual structures or processes of the ecosystem that underlie ES. Each ES has specific ecological requirements to be fulfilled to some minimum level (qualitative or quantitative) to enable functional delivery. For example, soil fertility (e.g. decomposition processes) requires different functional groups of soil biota, and the condition of each ecological requirement may be assessed by using proper indicators representing these functional groups. See Table 4-2 for the connections between ecological requirements and ES:

Table 4-2. Regulatory functions and other aspects of the ecosystem as ecological requirement underlying ecosystem services, from (Faber and Van Wensem, 2012).

Ecological requirement	Ecosystem service (clustered)					
	Soil fertility	**Adaptability, resilience**	**Buffer and reaction function**	**Biodiversity and habitat provision**	**Disease suppression and pest resistance**	**Physical structure**
Functional biodiversity	X	X	X	X	X	
Structural biodiversity (species richness)	X	X	X	X	X	
Ecosystem productivity	X	X		X	X	
Organic matter fragmentation, mineralisation	X		X	X		
Soil properties (pH, CEC, aggregates, pore space, WHC, etc.)	X		X	X		X
Nutrient cycling (supply, availability, assimilation, immobilisation)	X		X	X		
Autonomic development (nature)	X	X		X		
Soil organic matter build-up and maintenance	X		X		X	X
Carbon sequestration	X		X	X		
Greenhouse gases	X		X	X		
Groundwater supply and quality	X		X	X		X
Genetic variation and storage of genes		X	X	X	X	
Natural attenuation		X	X	X		
Adaptability, flexibility for use		X				
Air quality amelioration			X			
Water transport and storage			X			X
Landscape diversity				X		X
Soil archive (archaeological, geological)						X

In Thomsen et al. (2012), the ecological requirements shown in Table 4-2 are broadly grouped into 1) general biodiversity aspects, 2) microbial aspects, 3) plant aspects, 4) fauna aspects, 5) physical/chemical aspects – see Table 4-3.

Note: As defined in the terminology section, ecological requirements are effectively synonymous with soil functions.

- **Sub-problems level 3: ecological indicators** – indicators were designated as the means for assessing the state of ecological requirements; including indicators for soil biota, soil processes, or conditions of ecological sustainability, see Table 4-3. Numerous ecological indicators exist and may be used (as noted in previous sections). In this framework, preference was given to those that have been used in toxicity testing in the field or in the lab (i.e. thresholds based on no observed effect concentrations and SSD), and if so were assessed using ecotoxicity data.

Table 4-3. Sub-problems at level 1, 2 and 3 of the problem tree. Grey cells in the third column represent ecological indicators for which toxicological data are available for representative contaminants, adapted from (Thomsen et al., 2012).

Sub-problems at level 1: ecosystem service	**Sub-problems at level 2: ecological requirements**	**Sub-problems at level 3: ecological indicators**	
Soil fertility	General biodiversity aspects	1	Biodiversity indices
	Microbial aspects	2	Arginine deaminase activity
		3	Carbon sources utilisation diversity
		4	Cellulase activity
		5	Microbial biomass and activity
		6	Mycorrhizal infestation
		7	Nitrification
		8	Phosphatase activity
		9	Soil respiration
		10	Sulphur oxidation
		11	Urease activity
	Plant aspects	12	Dicotyledons biomass (fodder quality)
		13	N content (fodder quality)
		14	Litter standing crop
		15	Root density
		16	Root turnover
		17	Vegetation biomass
		18	Vegetation standing crop
	Fauna aspects	19	Anecic earthworms
		20	Ants
		21	Cattle meat quality
		22	Collembola
		23	Earthworm community structure
		24	Earthworms
		25	Epigeic earthworms
		26	Hoverflies, other dipterans, beetles
		27	Isopods
		28	Millipedes
		29	Mites
		30	Native bees
		31	Nematode community composition
		32	Nematodes
		33	Pollinators
		34	Protozoa
		35	Slugs, snails, beetles
		36	Springtails
		37	Springtails, mites
	Physical/chemical aspects	38	Ionic strength

		39	Loss on ignition
		40	Soil aggregates
		41	Soil bulk density
Adaptability and resilience	Fauna aspects	42	Earthworm ecological groups
		43	Mites functional groups
		44	Nematode community structure
		45	Oribatid mites
	General biodiversity aspects	46	Diversity indices
		47	Rank abundance distribution
		48	Soil food web complexity
	Microbial aspects	49	Fungi:bacteria ratio
		50	Nucleic acids microbial population characterisation
Buffer and reaction	Fauna aspects	51	Anecic earthworms
		52	Earthworm bioturbation activity
		53	Epigeic and endogeic earthworms
	Microbial aspects	54	Carbon sources utilisation diversity
		55	Methanotrophic diversity
	Physico-chemical aspects	56	Loss on ignition
		57	Soil organic matter
	Plant aspects	58	Litter standing crop
		59	Primary production
		60	Root turnover
		61	Tree growth
		62	Vegetation standing crop
Disease suppression	Fauna aspects	63	Predator species diversity
	General biodiversity aspects	64	Green vein landscape elements
		65	Key species
	Microbial aspects	66	Antibiotics producers
Biodiversity	General biodiversity aspects	67	Diversity indices
		68	Growth form diversity
		69	Isoenzymes
		70	Keystone species
		71	Species diversity
Physical support	Microbial aspects	72	Nucleic acids microbial population characterisation
	General biodiversity aspects	73	Vegetation cover
	Physico-chemical aspects	74	Soil aggregates
		75	Soil bulk density
		76	Soil stratification
	Plant aspects	77	Root density

Clearly, as noted by Thomsen et al. (2012), a critical issue is to find a reduced core set of the 77 ecological indicators listed Table 4-3, using e.g. a logical sieve approach (Faber et al., 2013; Ritz et al., 2009; Stone et al., 2016b). To create an indicator 'shortlist', the authors recommend to first group the ecological indicators in clusters of similar importance (i.e. equivalent weights) for types of ES relevant to certain land uses, then assess the data coverage for the most important ecological indicators. Their classification method (ranking the indicators according to the derivation of soil quality parameters using toxicological data to fulfil the defined ecological requirements) showed that for the 3 types of land uses in focus, no ecological indicator relevant to all 3 is covered with toxicological data. This indicates a gap in the data and uncertainty due to ignorance concerning how an ecological requirement is affected. When urban recreational greenspace is deemed less important than agricultural or natural uses then the 8 ecological indicators covered with toxicological data for chemical stressors, shown shaded in grey in Table 4-3, become more applicable. Regarding uncertainty in data and understanding of what actually contributes to ecosystem functioning, a pre-cautionary approach could be used to define maximum ecological productivity by appreciating some level of soil ecosystem requirements is needed for maintaining ecological integrity. Furthermore, knowledge of soil ecosystem requirements may guide the planning of health improving soil management practices (Thomsen et al., 2012).

Ecosystem services, their ecological requirements and indicators can be ranked or weighed by either societal or ecological importance with respect to a specific type of land use (Faber and Van Wensem, 2012). This will likely affect the weighing of assessment results, thus the outcome of decision-making, but would enable accounting for a societal viewpoint as stakeholders may attach different values to ES and their indicators (Faber and Van Wensem, 2012). Within an ERA, indicators may also be ranked according to increasing ecological relevancy as they differ in relevancy for the various ES (Faber and Van Wensem, 2012).

To summarise, the method by Thomsen et al. (2012) (and discussed by (Faber et al., 2013; Faber and Van Wensem, 2012)) presented a tentative classification of soil health criteria in terms of their relevance to land use, serving as a basis to recognise vulnerable and sensitive structures and processes in ecosystems that are needed in the provision of ES. Also, this framework provides clear, robust linkages between land uses, ecosystem services, ecological requirements and indicators as shown in Figure 4-4 and Figure 4-5. Terminology may differ slightly (e.g. ecological functions vs. soil functions), however, the framework aligns closely with the understanding of the relationships between assemblages of soil biota, soil processes, ecosystem/soil functions and the ultimate delivery of ES as discussed throughout this review and shown in Figure 2-5 and Figure 2-4.

Gutierrez et al. (2015) – modified Triad tiered approach

The TRIAD approach combines three 'Lines of Evidence' (LoE) for the estimation of ecological risk into an 'integrated risk value' (IRV) that provides a clear indication of the ecological risks at a site (ISO, 2017b; Mesman et al., 2006). The 3 LoE include 1) *chemical* (total and bioavailable contaminant concentrations), 2) *toxicological* (a variety of toxicity bioassays), and 3) *ecological* (parameters providing information on the abundance, activity and diversity of soil organisms). The analysed parameters are further separated for the three LoE into 3 tiers of increasing in complexity and cost. The risk assessment then proceeds according to 5 steps shown in Figure 4-6.

- **Tier 1** (screening) – cheaper and quicker tests (e.g. total concentrations)
- **Tier 2** (refining) – more specific and complex tests
- **Tier 3** (detailing) – most complex and site-specific

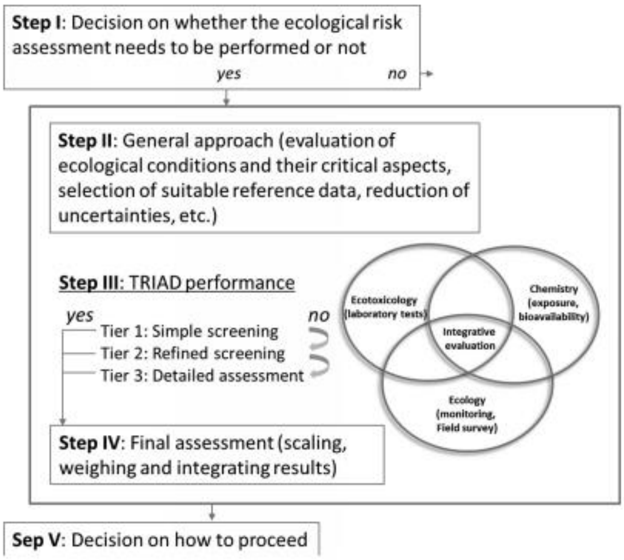


Figure 4-6. Decision tree of the five steps of integrating soil quality TRIAD into a site-specific ERA, from (Ok et al., 2020) *(modified after* (ISO, 2017b)*).*

The ERA methodology proposed by Gutierrez et al. (2015) is intended to be site-specific and further takes into consideration the ecosystem services to be protected that result from ecological functions attributable to the concerted actions of a myriad of soil organisms. In turn, the selected soil ES will depend on the envisioned land use for the study site, see Table 4-4, which were classified in accordance with the Millennium Ecosystem Assessment (Gutiérrez et al., 2015; Millennium Ecosystem Assessment, 2005).

Table 4-4. Ecosystem services for each land use, adapted from (Gutiérrez et al., 2015).

	Natural	Agricultural	Urban	Industrial
Supply of nutrients	x	x		
Regulation of water cycle	x	x	x	x
Pest control		x		
Regulation of carbon flux and climate control	x	x	x	
Decontamination and bioremediation		x	x	x

This methodology is carried out in 6 broad steps, shown in Figure 4-7 and described below:

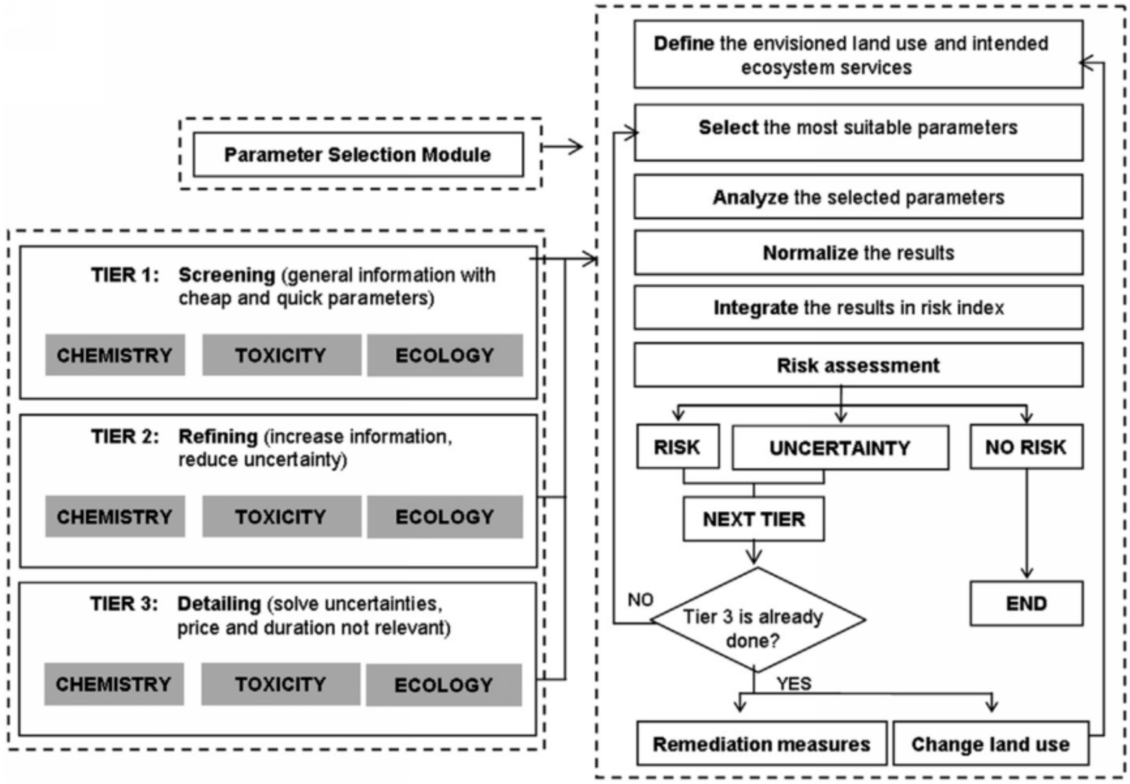


Figure 4-7. Methodological scheme for the contaminated site-specific TRIAD-tiered ERA approach, from (Gutiérrez et al., 2015).

1. **Definition of land use, intended ecosystem services and nature of contaminants** – as shown in Table 4-4, the ERA is tailored to protect specific ES depending on the envisioned land use. Linking the measured parameters to ES and land use objectives (i.e. stakeholder demands) enhances the societal relevance of the ERA, hence increasing acceptance by decision-makers. The 'nature of contaminants' refers to which type of contaminants are present at the site (e.g. organic or inorganic) and how this could affect the selection of suitable parameters.

2. **Selection of suitable parameters in 'Parameter Selection Module' (PSM)** – the PSM was developed to aid in the selection of suitable parameters (i.e. indicators) to use in the ERA depending on the site-specific conditions considering envisioned land use, intended ES and the nature of the contaminants. Based on a multi-criteria analysis (see (Critto et al., 2007; Semenzin et al., 2009b, 2007) for more details on the method) each parameter was evaluated according to criteria selected for each LoE: chemical, toxicological and ecological. 'Importance values' (prioritisation values) were calculated from average scores obtained during the expert judgement process. Then, all parameters were numerically scored by using a scoring equation and then ranked within each tier. In total, 3 chemical, 12 toxicological and 15 ecological parameters were included in the PSM for scoring.

   Note: The PSM is comparable to SQI selection using a multi-criteria logical sieve approach.

   Biological parameters (i.e. bioindicators) were emphasised for inclusion in the PSM, especially microbial parameters since they are responsible for 80-90% of the soil biological activity and have been successfully used to determine the impact of metal contamination on soil quality as well as to assess the effectiveness of (gentle) remediation techniques applied to metal contaminated soils (Epelde et al., 2008, 2009b, 2010; Gómez-Sagasti et al., 2012; Gutiérrez et al., 2015). Earthworms were also highly valued due to their function as ecosystem engineers, ability to accumulate many xenobiotics and mix soil to facilitate organic matter cycling (Gutiérrez et al., 2015).

3. **Determination of parameters within each LoE** – accounting for both the envisioned land use and the nature of the contaminants at the site, the most suitable parameters for each tier can be selected using expert judgement and the results of scoring from the PSM. **Generally, less expensive, faster and highly standardised parameters ranked highly for tier 1, but become less important for tiers 2 and 3 where greater endpoint and ecosystem sensitivity were valued** (Gutiérrez et al., 2015). Selection in their case study also accounted for preference in parameters depending upon their relevance. For example, soil basal respiration and microbial biomass carbon (MBC) were favoured for tier 1, because basal respiration is a well-known indicator of overall microbial activity and MBC, despite its lower score in the PSM, due to its utilisation in a variety of quotients traditionally used as SQI (e.g. respiratory quotient – qCO_2 and specific enzyme activities). Upon selection, the parameters can be performed for the site in question according to established methods.
4. **Normalisation of results** –the data from the measured parameters need to be normalised to a common scale, representing 'ecosystem impairment', in order to make them comparable and integrable in a 'risk index', and to facilitate interpretation and decision-making processes. The ecosystem impairment scale ranged from zero to one (0 = no effect; 1 = maximum impairment), which was based off Semenzin et al. (2008). A control (non-contaminated) soil is set to an impairment of 0; then, ecosystem impairment for the different sampling points is calculated by means of a linear function (from 0 to 1), considering the corresponding value of the control soil for each parameter as zero ecosystem impairment.
5. **Integration of results in risk index** – as is frequently done in an ERA based on the TRIAD approach, the integration and subsequent interpretation of the data into an 'integrated risk index' (IRI) is required in order to get a scientific evidence-based risk assessment (Gutiérrez et al., 2015). Thus, the IRI is considered as the final outcome of the 'Weight of Evidence' approach (Gutiérrez et al., 2015). Risk indices facilitate decision-making and risk communication; however, risk indices might oversimplify the complexity of site-specific risks and the assessment itself (Gutiérrez et al., 2015; Semenzin et al., 2007). For the calculation of quantitative risk indices, ecosystem impairment values were first integrated in the 3 LoE-risk indices using mean values, and then the IRI was calculated by averaging the 3 LoE-risk indices.
6. **Risk assessment** – interpreting the IRI was done according to the ranges shown in Figure 4-8. The risk magnitude, and consequent outcome of the ERA, depends on the IRI value as well as on the level of uncertainty given by the standard deviation (SD) of the values, while also taking into consideration the envisioned land use (Gutiérrez et al., 2015; Mesman et al., 2006). In practical terms, this means that if an elevated risk for the soil ecosystem is detected after Tier 1 assessment or the level of uncertainty is unacceptable, the ERA must proceed to Tiers 2 and 3, using more specific, complicated and costly tests (Gutiérrez et al., 2015). If an ecological risk is still detected after Tier 3 assessment then remedial actions must be employed or, alternatively, a different, less restrictive land use must be envisioned (Gutiérrez et al., 2015).

SD ≤ 0.4	Integrated risk index		0.00–0.20	0.21–0.50	0.51–0.75	0.76–1.00
	Risk assessment	Quantitative	No risk	Low	Moderate	High
		Qualitative	No evidence	Slight evidence	Moderate evidence	Strong evidence
	Acceptable land use		N, A, U, I	A, U, I	U[1], I	I[1]
SD > 0.4	Integrated risk index		0.00–0.20	0.21–0.75		0.76–1.00
	Risk assessment	Quantitative	Low	Moderate		High
		Qualitative	Slight evidence	Moderate evidence		Strong evidence
	Acceptable land use		A[2], U, I	U[1], I		I[1]

Figure 4-8. Quantitative and qualitative risk assessment depending on integrated risk indices (IRI) and standard deviations (SD). Acceptable land uses are indicated: N = Natural, A = Agricultural, U = Urban, I = Industrial. (1) Only acceptable in case of sealed soil, with no green bared areas; (2) An assessment of the target of concern is recommended, from (Gutiérrez et al., 2015).

To summarise, the modified TRIAD-tiered approach of Gutierrez et al. (2015) provides a robust, stepwise procedure for accounting for and assessing the delivery of ES in the context of potential impairment due to contamination.

Integrative numerical index

As shown in the example with Gutierrez et al. (2015), integrative indices can be used in an ERA approach. Other approaches that focus more specifically on soil quality assessment have also uses indices to synthesise the information collected from a range of soil physical, chemical and biological parameters into a 'soil quality score' (Turbé et al., 2010). According to Turbé et al. (2010), the main advantage of this kind of the numerical, index-based approach is that it simplifies interpretations to allow comparisons between different soils. The index-based approach is also advantageous when interpreting the data in terms of ecosystem services to communicate with land owners and decision-makers regarding the state of soil resources as a habitat for soil organisms and for human well-being (Pulleman et al., 2012).

One prominent example of this approach is the General Indicator of Soil Quality (GISQ) method (discussed previously in Section 3) which derives a synthetic numerical indicator for soil quality based upon five compound sub-indicators representing five ecosystem services through a number of physical, chemical and biological parameters by using statistical multivariate analyses (Velasquez et al., 2007). This statistical approach can be universally applied by calibrating to the site context and allows for monitoring of change through time and variation between sites without relying on expert opinion (Pulleman et al., 2012; Velasquez et al., 2007)

Another example, previously discussed in Section 3, is the soil quality index developed by Epelde et al. (2014b) to statistically calculate the overall soil quality (by grouping soil microbial parameters within a set of ecosystem attributes) using Equation (1) below:

$$ES = 10^{logm + \frac{\sum_{i=1}^{n} |logn_i - logm|}{n}} \qquad (1)$$

Where, *m* is the control value (set to 100%) and *n* corresponds to the measured values for each parameter as a percentage of the control value (Epelde et al., 2014b). Epelde et al. (2014b) conclude that this index is appropriate for the assessment of soil quality in those cases where the soil has been intentionally treated to increase some parameters, e.g. addition of amendments or plants to remediate soil. This was expanded upon in Burges et al. (2016, 2017) to test the validity of such an approach to evaluate the effectiveness of phytoremediation studies. The soil physicochemical and biological indicators were grouped within a set of 8 ecosystem services as follows: 1) *nutrient cycling* – enzyme activities and basal respiration, 2) *carbon storage* – total C content and MBC, 3) *water flow regulation* – water content at field capacity, 4) *water purification* – pH and $CaCl_2$-extractable metal concentrations, 5) *contamination control* – pH, $CaCl_2$-extractable and total metal concentrations, 6) *pest control* – soil suppressiveness, 7) *fertility maintenance* – total N and organic C content, and 8) *biodiversity* – richness (S), Shannon's diversity (H') and Pielou's evenness (J') of bacterial and fungal communities from ARISA profiles. The effects of the treatments on soil quality were then assessed according to an integrated numerical index for each ecosystem service, shown below in Figure 4-9 for an example project.

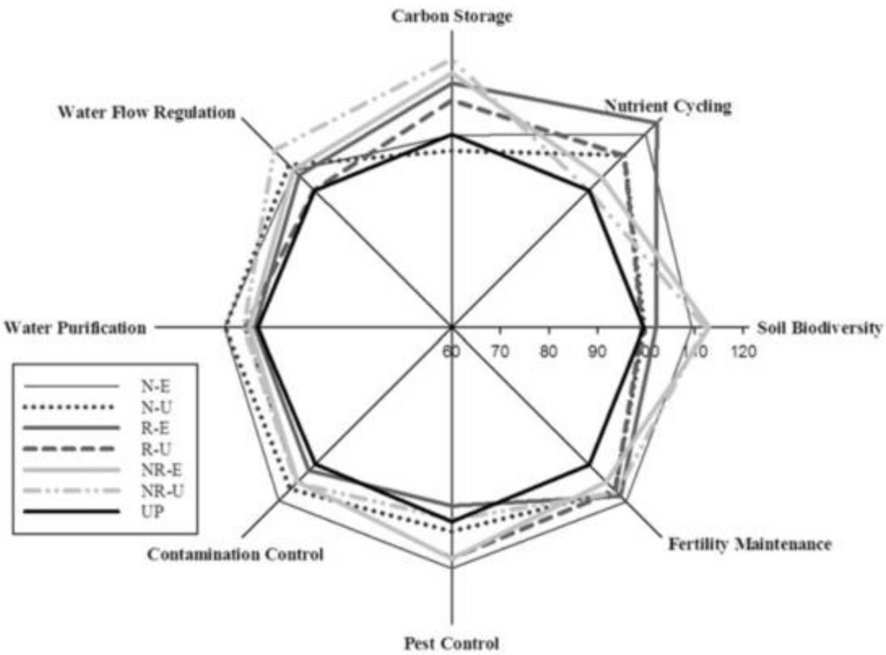


Figure 4-9. Sunray plot of ecosystem services, from (Burges et al., 2017)*. A value of 100 corresponds to the mean value obtained for each ecosystem service in the unplanted (UP) control treatment. N-E, N-U, R-E, R-U, NR-E and NR-U refer to the various phytoremediation treatments using different plants with or without an endophyte consortium, see* (Burges et al., 2017) *for more details.*

The authors conclude that the assessment of soil quality through the grouping of soil parameters in ecosystem services was an appropriate approach to estimate the effectiveness of phytoremediation (Burges et al., 2017, 2016), **especially considering that the ultimate goal of any soil remediation method must not be only to remove the contaminants (or instead disrupt source-pathway-receptor linkages) but also to restore soil quality** (Epelde et al., 2008; FAO et al., 2020; Gómez-Sagasti et al., 2012). Soil microbial parameters were in focus in these studies as they have been shown to have great potential as biological indicators of the effectiveness of phytomanagement and recovery of soil quality (Epelde et al., 2009a; Garbisu et al., 2011; Gómez-Sagasti et al., 2012). Microbial parameters, however, can be highly context-dependent and difficult to interpret thus grouping them into higher-level categories such as ecosystem services will facilitate interpretation and decision-making as well as provide long-term phytomanagement monitoring programs with the ability to adapt through time against changes in techniques, methods, interests, etc. (Burges et al., 2018, 2017, 2016; Epelde et al., 2014a; Garbisu et al., 2011; Gómez-Sagasti et al., 2012).

4.1.2 Ecosystem service mapping

Within the context of contaminated sites, methods to evaluate existing ecosystem services and the expected changes due to a remedial action have emerged in recent years. Scopus search results for "brownfield" (and related terms) AND "ecosystem services" showed 196 hits of which many were relevant. However, "brownfield" AND "ecosystem service mapping" showed only 5 hits, 2 of which were deemed relevant. A few of these studies will be discussed in this section.

In the Swedish context, the Swedish EPA (SEPA) created the *Guide to valuing ecosystem services* to meet the growing interest and demand for valuing ES in a variety of applications, including contaminated sites (SEPA, 2018). The guide focuses on methodological aspects of valuation emphasising the importance of methodology to ensure that important values are not missed and that the information required to make a proper valuation is generated (SEPA, 2018). Another important aspect is that ecosystem service values are not always expressed in monetary terms, discussed in more detail in the following section, but can be alternatively expressed by means of words and description (qualitatively), by means of scoring (semi-quantitatively), or in the form of various other physical units (quantitatively) (SEPA, 2018). Broadly speaking, SEPA breaks down the valuation of ecosystem services into six steps (SEPA, 2018):

1. Identifying the purpose of the valuation,
2. Identifying ecosystem services (e.g. a broad list of potentially relevant ES),
3. Defining the analysis (i.e. limiting the scope),
4. Determining the starting points for the valuation (i.e. investigate the ES value creation chain),
5. Apply valuation methods (i.e. qualitative, semi-quantitative, quantitative or monetary),
6. Doing a review.

Semi-quantitative and qualitative valuation

One promising method, mentioned in SEPA's guidance material and aligning closely with the stepwise methodology, assesses the sustainability of different remediation alternatives via a semi-quantitative (i.e. quantifying values by assigning them points between 1-5) "ecosystem service mapping" procedure[9] (SEPA, 2018). Initially developed by Ivarsson (2015) as part of the *Balance 4P* project (Norrman et al., 2015) and expanded upon by Volchko et al. (2020) in the *Applicera* project (as part of a broader cost-benefit analysis), the analysis is performed in the following steps (when considering uncertainties as in (Volchko et al., 2020)):

1. Screening a gross list of ES then identifying the relevant ES and their current status on the site given present land use (i.e. reference alternative),
2. Assigning a degree of importance to each ES based on current demand,
3. Assigning a score between 1 and 5 (very good and very limited, respectively) which reflects the capacity of a site to deliver a particular ES, while also specifying a level of uncertainty for the provision of this ES (low, medium, high),
4. Evaluation of the effects of the remediation alternatives on each ES relative to the reference alternative regarding changes in quality and quantity,
5. Summing the effects on each ES while accounting for the demand for each ES.

The outcome of this procedure would be a comparison of the remediation alternatives according to expected positive or negative impacts resulting from each alternative on the relevant urban

[9] Understandably, confusion arises at the use of the term 'mapping' which may or may not include spatial (e.g. GIS) mapping. In the case of Ivarsson (2015) and Volchko et al. (2020) it does not but the use of the term depends on the individual study.

and soil-based ecosystem services, shown in Table 4-5. By applying a semi-quantitative approach (i.e. evaluating indicators on a point scale between 1-5), the changes in ecosystem services resulting from various remediation alternatives could be evaluated (Ivarsson, 2015). The value in this method is that it brings urban and soil ES to the forefront of the decision-making process in a clear, scalable way by providing indicators (or proxy indicators) for quantifying and evaluating the expected change in the services for each alternative (Ivarsson, 2015). The list of indicators used to measure each ecosystem service is shown in Appendix III.

Table 4-5. List of ecosystem services included in the ecosystem service mapping assessment, adapted from (Ivarsson, 2015)

Type	Category	Ecosystem Service
Urban	Provisioning	Food, fresh water
	Regulatory	Air quality, climate (global), climate (local), water, noise reduction, water purification and waste treatment, pollination and seed dispersal, maintaining nursery populations and habitats, natural hazard regulation
	Cultural	Knowledge systems, aesthetic values, cultural heritage, recreation
Soil	Provisioning	Food, biomass
	Regulatory	Water purification, climate regulation (global), water regulation, erosion regulation, waste treatment

An expanded version of the ES mapping methodology paired with cost-benefit analysis (CBA) was recently applied as part of the *Applicera* project for improving ecological risk assessment in Sweden, see (Volchko et al., 2020). Important differences from the original method include the alignment of the gross ES list to be screened in step 1 with the CICES classification system and SEPA framework (seeAppendix III), integration within a broader socio-economic analysis (i.e. CBA) of remediation alternatives and a probability-based uncertainty and sensitivity analysis of the expected changes in ES due to a remedial action. Altogether the methodology used in *Applicera* proceeded according to the steps shown below in Figure 4-10:

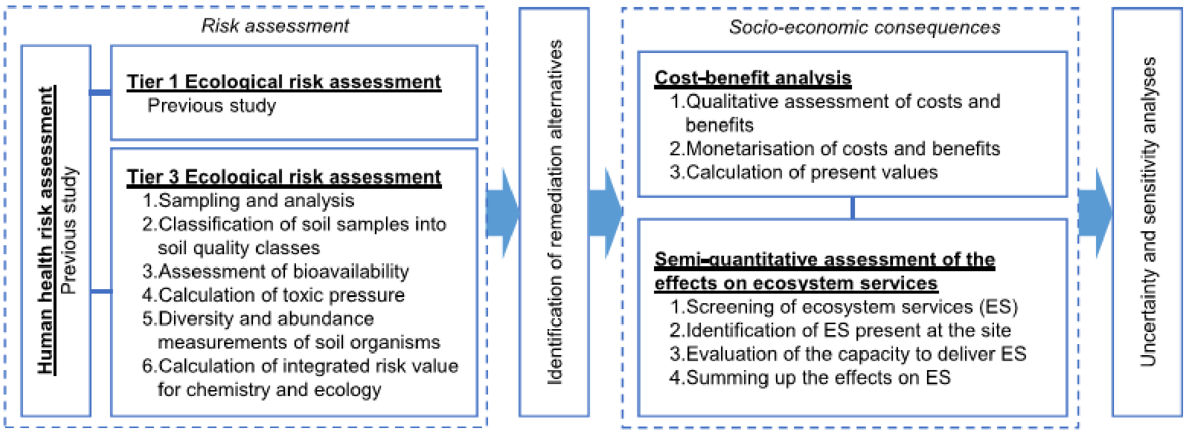


Figure 4-10. The methods used in the Applicera study including the ES mapping procedure, from (Volchko et al., 2020).

Other examples of an ES mapping approach include the study by Cortinovis and Geneletti (2018) who used spatial mapping techniques (i.e. GIS) as a tool to prioritise development at urban brownfield sites in order to provide the two key ES of microclimate regulation and nature-based recreation. Their method adopted a semi-quantitative multi-criteria analysis using weights for various parameters reflecting the different planning and beneficiary group objectives that was then used to inform upon which site could potentially provide the greatest benefits in terms of ES provisioning (Cortinovis and Geneletti, 2018).

Pueffel et al. (2018) provide another illustrative example evaluating specifically cultural and recreational ecosystem services (according to CICES classification) at green brownfield sites (i.e. abandoned sites primarily covered by vegetation) by mapping the use of various brownfield sites via a smartphone application (MapNat) (Pueffel et al., 2018). Based on citizen responses, they identified spatial use patterns in how the urban green brownfields in Leipzig were used and showed that such sites were still highly valued for recreational services such as dog walking, exercise or informal visits. The researchers noted that, for example, the form of ES varies with the site characteristics and even sites with low accessibility (due to e.g. fencing or remoteness) were highly valued especially if they had a pleasing natural 'urban wilderness' character with relatively rich and diverse vegetation (Pueffel et al., 2018).

Mathey et al. (2015) took a different approach and applied three different methods to determine the extent of three selected ecosystem services provided by green urban brownfields (i.e. more or less vegetated brownfield sites): 1) identification of parameters for habitat services based on a literature review, 2) modelling of microclimatic effects for microclimate regulating services at different scales (site level and city level), and 3) a qualitative survey on the perception, acceptance, and use of green urban brownfields for recreational services which was expanded on in Mathey et al. 2018. Furthermore, these specific results were used to evaluate the impacts of land use changes and the potentials of design options of urban brownfields on the provision of ecosystem services. This was based on land-use scenarios and the qualitative evaluation of eight design options for green space development on urban brownfields (Mathey et al., 2015), see Figure 4-11.

Options for Green Spaces on Brownfields	Habitat Services	Microclimate Regulation Services	Recreational Services
Urban Agriculture	+/-	+ / ++	+/-
Urban Woodland	+ / ++	++	+ / ++
Gardens	+/-	++	++
Sports/Leisure Pursuits	+/-	+	++
Venues for Cultural Projects/Public Events	-	+	+
Low-Intervention Park	++	++	+ / ++
Nature Experience Areas	++	++	++
Urban Wilderness	++	+ / ++	+/-

Figure 4-11. Options for reusing brownfields as green space and evaluation of their habitat services, microclimatic regulation services, and recreational services: ++ "well suited", + "suited", - "unsuited, +/- "detailed investigation of individual site necessary", from (Mathey et al., 2015).

Quantitative valuation

De Valck et al. (2019) adopt a slightly different tack to value the urban ecosystem services (UES) provided by green infrastructure at brownfield sites. Five UES (local climate regulation, air filtration and ventilation, recreation, carbon sequestration and avoided runoff) were quantified and economically valued using the 'Nature Value Explorer' (Nature Value Explorer (natuurwaardeverkenner.be)) modelling tool for a case study site in Antwerp where three types of green infrastructure were employed: a green corridor, infiltration gullies and green roofs. Using the valuation metrics shown in Table 4-6, the authors were able to monetise the expected 'biophysical flows' delivered by the three types of green infrastructure on each UES. Altogether, green infrastructure at the site would generate an estimated €700,000/year (€19,069/ha*yr) (De Valck et al., 2019).

Table 4-6. Monetary valuation factors used to quantify urban ecosystem services of green infrastructure on-site, adapted from (De Valck et al., 2019).

Urban ecosystem service	Metric	Value per unit (€)
Avoided runoff	m^3/year	0.52
Air filtration and ventilation	kg PM10/year	72
Climate regulation	m^2/year	0.207
Carbon sequestration	tons/year	233
Recreation	visitors/year	4.5

While the authors acknowledge that this methodology has a number of limitations (e.g. dependent upon the values per unit of UEC obtained from literature study, different calculation methods for recreation, difficulties in accurately modelling effects from vegetation), accounting for UES provides valuable decision-support alongside project costs and economic return when comparing different potential brownfield redevelopment designs. The advantages of this UES valuation approach are that it is relatively simple to apply and effective at demonstrating the many benefits of green infrastructure in easily understandable monetary terms (De Valck et al., 2019).

Applicability

The most suitable use of these ES valuation schemes in the context of contaminated soil and land management is to provide support in selecting the most suitable remediation alternative or to justify a brownfield regeneration project in an urban area. Furthermore, by mapping the use of brownfield land the needs and desires of local citizenry can be accounted for and used as an indicator to inform land management and planning (Mathey et al., 2018, 2015; Pueffel et al., 2018). These types of ES mapping and valuation methods could be included in existing brownfield re-development decision-support tools that do not yet account for them since ES considerations are a valuable addition to any sustainability-minded decision-making and may greatly help urban planners and land managers to mitigate undesirable environmental impacts (De Valck et al., 2019; Mathey et al., 2015).

As stated by Ivarsson (Ivarsson, 2015), the ES mapping method '*indicates that a semi-quantitative approach to map the changes in provision of ecosystem services that will follow from different redevelopment alternatives will potentially add important decision support regarding the economic and social desirability of available options. The principal strength of the method is its ability to map and quantify changes in well-being that in many cases are neglected in applications of cost-benefit analysis to redevelopment projects, despite the relevance of those changes in such analysis.*'

4.2 Economic valuation

The necessity of ecosystem service assessment to demonstrate the value of ES for society at large has been illustrated well in the Swedish context in the report "Make the value of ecosystem services visible" (SOU, 2013) and SEPA's "Guide to valuing ecosystem services" (SEPA, 2018), which both highlight the necessity of clear valuation of ES for sound decision-making and ensuring that the long-term needs of society for functioning ecosystems are met. When discussing the valuation of ecosystem services, monetary valuation (i.e. measuring the value of an ES in monetary units) is typically the foremost considered valuation method (SEPA, 2018; TEEB, 2011, 2010). However, there are multiple other ways other than monetisation to value ecosystem services; including, by means of words and description (qualitatively), by means of scoring (semi-quantitatively, as in the ES mapping method), or in the form of various other physical units (quantitatively) (SEPA, 2018; TEEB, 2011, 2010). In fact, monetisation is considered to be challenging to account for most ES (Söderqvist et al., 2015; Volchko et al., 2020). For, in order to be able to compare various remedial actions and their effects on the environment, one must be able to value how ecosystem services change with the implementation of different site remediation actions, including comparing the necessary trade-offs (Colombo et al., 2012; SEPA, 2018). In the case of incorporating ES in a CBA, ES values are best quantified in monetary terms (monetisation) to include as quantified ecosystem service metrics that would facilitate evaluating costs and benefits (SEPA, 2018). This would entail that the value of an ecosystem service is described using on or more indicators for the service that are measurable aspects of the environment that contribute to human well-being for which a monetary value can be credited (SEPA, 2018). In general, there are three main categories for monetary valuation of ES: 1) *Market-based valuation* – based on services that are marketized with affixed prices or expenditures for protection from contaminated water, noise, flooding, etc.; 2) *Non-market valuation* – including 'scenario valuation methods' creating hypothetical or surrogate markets in which participants indicate their 'willingness-to-pay' for certain services via surveys (stated preferences) or 'market data methods' based on studying the relationships between ecosystems and actual behaviours according to market prices and production data (revealed preferences); and 3) *Value transfer* – provides an approximation of an ES's value by generalising a value from a study done in an unrelated place (Baveye et al., 2016; SEPA, 2018; TEEB, 2011, 2010). However, there are pros and cons with each of these valuation methods; for example, the willingness-to-pay for species or measures that are unfamiliar or undesired by the general public could yield extremely low values despite the fact that these species could perform indispensable ecological services (Schröder et al., 2018). See (Chee, 2004) and (Baveye et al., 2016) (Figure 4-12) for more in-depth reviews of the many and varied methods that can be used to value ecosystem services within the neoclassical economics market framework.

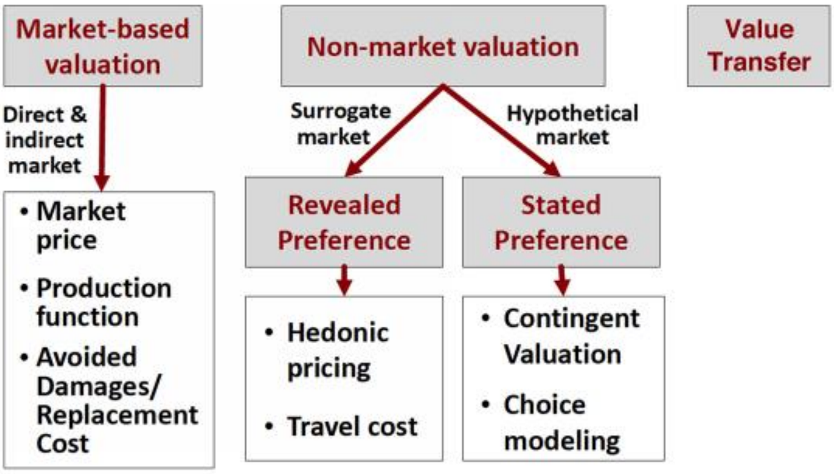


Figure 4-12. Schematic classification of the different methods that have been developed for the monetary valuation of ecosystem services, from (Baveye et al., 2016)

The appropriate method to valuate ecological goods and services (i.e. ecosystem services, ES) is a controversial subject with no clear answer other than that ES should be included in some form in economic valuations such as cost-benefit analyses. One such debate on the topic of monetary valuation (monetisation) in ecological valuation can be read in Society for Ecotoxicology and Chemistry's 'Learned Discourses' special issue on ecological valuation

(Chapman et al., 2015). Three distinct views are discussed: 1) counting money in, 2) including non-monetary means of valuing ES, or 3) using money when possible though it is not always possible. The arguments range from Calow (2015) stating that monetisation is crucial and (aligning with the philosophy of cost-benefit analysis) market values are the only way to ensure transparency in capturing public preferences for ES (so-called *revealed preference* (Johansson and Kriström, 2018)) to Kapustka and McCormick (2015) arguing that neoclassical economics is fundamentally flawed and monetisation is insufficient to capture the value of ecosystems (often setting aside important parts in calculations as 'externalities') thus we must transition to more eco-centric economic models. Ultimately, the middle way is best stated by Munns and Rea (2015) by saying that the ES concept is fundamentally anthropocentric whose value is defined 'in the eye of the beholder' (i.e. people and society) and money is a convenient 'common unit' with which to quantify, aggregate, and compare these values in the decision-making process; however, monetisation is not always feasible, practical, not desirable.

Other examples include the work by Dominati et al. (2010) who argue that pairing their model with an economic valuation of soil services provides powerful decision-support to economics and policy makers in order to better understand and value the ecosystem services provided by soils in development. This model predicates a valuation based on natural capital accounting, which is supported by other work to economically value ecosystem services (Dominati et al., 2010; Kelemen et al., 2015; Maes et al., 2016; Robinson et al., 2013), though it does pose a risk that indirect uses or non-use values may be neglected. To account for this, the term 'total economic value' (TEV) has been used to ascertain the total economic value generated by an ecosystem service by summing the use and non-use values (SEPA, 2018), as shown in Figure 4-13. According to Baveye et al. (2016), TEV is typically separated into 'extrinsic' or 'instrumental' value based on use or function and 'intrinsic' or 'inherent' value in the absence of direct use value though the notion of intrinsic value is controversial. Most often, the intrinsic value relates an 'aesthetic' or 'moral' (e.g. 'right to exist') value or to functions that result in benefits to nature (i.e. not to humans), and the criticism arises when attempting to incorporate these values into an anthropocentric, economic valuation (Baveye et al., 2016). Instrumental value is customarily divided into 'use value' and 'non-use' or 'passive use value' that are based on whether goods and services are interacted with 'directly', with further sub-groups added in specific cases (Baveye et al., 2016).

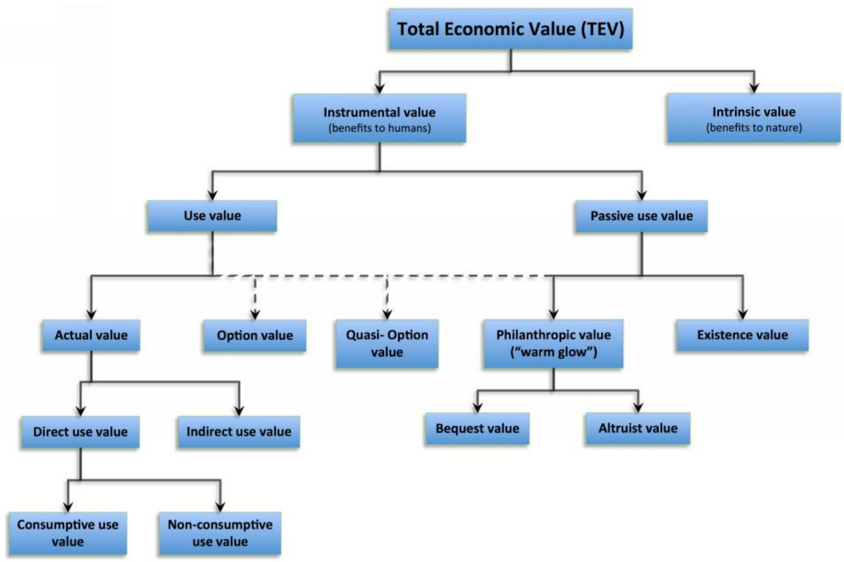


Figure 4-13. Schematic diagram of the subdivision of the Total Economic Value (TEV) into different types of value, from (Baveye et al., 2016).

Pascual et al. (2015) provide an alternative framework for the TEV of soil biodiversity and ES by joining the fields of soil ecology and ecological economics, see Figure 4-14. The authors contend that soil biodiversity and associated ecosystem services can be valued as a natural capital asset via two main additive value components, in the context of risk and uncertainty: a Total Output Value (TOV) and a Natural Insurance Value (NIV). The TOV refers to the different economic outputs associated with tangible benefits provided by soils in a given state, which is largely dependent on its 'use value.' The use value is separated into i) *direct use value* (provisioning services) – 'final' products of soils that can be used directly or 'consumed', ii) *indirect use value* (intermediate regulating services) – services necessary for the production of final services and which are typically valued by society as a whole but rarely fully captured, and iii) *option value* – referring to keeping soil healthy and viable for future purposes. Direct use value is also connected to *cultural services* as diverse ecosystem 'co-produce' cultural services, which may be intangible and difficult to value linking to 'non-use value' like *existence value*. NIV is related to the capacity of soil biodiversity to maintain the production of ecosystem services over time in the face of risk and uncertainty, which is linked to the idea of socio-ecological resilience. The NIV aims to account for the importance of regulating and supporting ES given fluctuating disturbance factors influencing the provisioning services under global environmental changes such as intensification of land use and climate change. This value comprises two value related components under risk and uncertainty: 1) *self-protection* – the value of lowering the 'risk' (probability) of being negatively affected by a disturbance (e.g. pests, flood, drought, etc.) which would decrease the mean value of the flow of an ES, and 2) *self-insurance* – the value of lowering the 'size of the loss' due to such an event occurring and is associated with the ecological notion of resistance. In essence, the NIV of soil biodiversity is associated with the stabilisation of the total output value of soil biodiversity and is assumed positive when the beneficiaries of such services are risk averse (Pascual et al., 2015).

The proposed framework emphasises regulating (e.g. water regulation) and supporting (e.g. nutrient cycling) services as a simpler way of identifying the value of soil biodiversity that can be considered as 'intermediate services' (indirect use) underlying 'final' (direct use) provisioning services thus avoiding double counting. In this context, soil biodiversity can be considered as a *portfolio of resources that build up soil natural capital which in turn can be economically valued.* Thusly viewed as an economic asset, the flow of ecosystem services derived from soil biodiversity is the accrued interest or return (positive or negative) from managing the asset. The authors further expound on the value of soil biodiversity when seen through this economic lens by incorporating some of the key contributions of soil biota to ecosystem services from a soil ecology perspective. In their examples, the crucial contributions of earthworms for the delivery of regulating services (water infiltration and greenhouse gas control) and provisioning services (grass production) are highlighted as being amenable to economic valuation within the proposed framework by using a proxy indicator (i.e. abundance and species type) to predict the delivery of the ES based on the 'stock' of soil biodiversity. Examples of increasing the NIV of soils would be the 'suppressiveness' (i.e. resistance and resilience) of a biodiverse soil to specific soil-borne pests diseases due to e.g. robust soil microbial biomass and communities that would suppress roots infections by pathogens through competition and/or antibiosis and rebound faster after disturbance (Pascual et al., 2015).

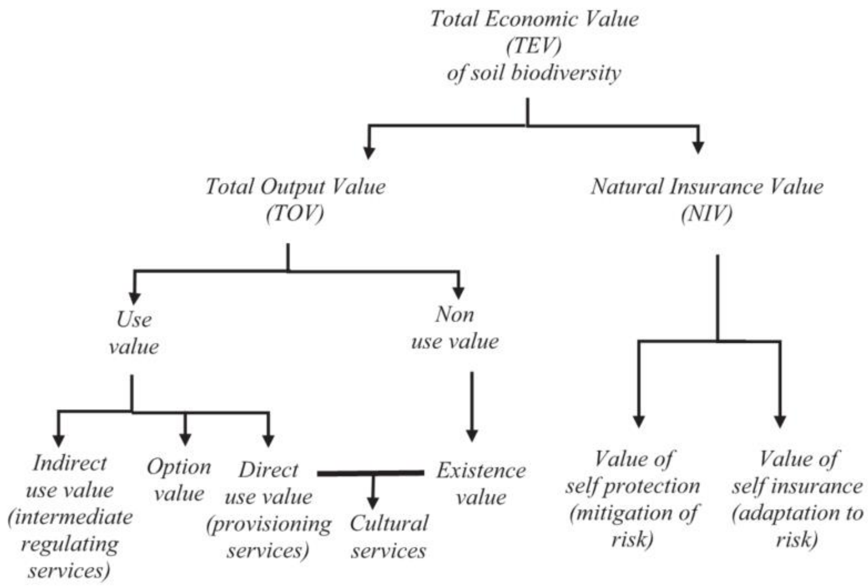


Figure 4-14. The Total Economic Value (TEC) of soil biodiversity based on the two main additive components of Total Output Value (TOV) and Natural Insurance Value (NIV) under risk and uncertainty. The lowest level value components are the ones that are most closely related to soil as natural capital in this framework. Cultural services are co-produced and jointly related to use and non-use value, from (Pascual et al., 2015).

According to Baveye et al. (2016), a prerequisite to progress in such public deliberations on the valuation of ES is that participants be cognizant of the extreme relevance of soils to many aspects of their daily life, and as long as this prerequisite is satisfied, the combination of deliberative decision-making methods with a sound scientific approach to the quantification of soil functions/services (including uncertainties) is a very promising avenue to manage, effectively and ethically, the priceless heritage that soils constitute. In addition, when certain ES are highly valued but not necessarily linked to immediate benefit for the landowner (e.g. climate regulation services), economic incentives like payment schemes for ecosystem services (PES), which aim to internalise the value of services provided by intact ecosystems into economies via markets, can be useful tools (Baveye et al., 2016; Pascual et al., 2015; Schomers and Matzdorf, 2013). According to Schomers and Matzdorf (2013), PES is a multi-facetted term with many diverse definitions coexisting, but a seminal definition is given by Wunder (2005) focusing on market transactions and construing PES as i) a voluntary transaction where ii) a well-defined ES (or a land-use likely to provide that service) iii) is being ‘bought’ by a (minimum one) ES buyer iv) from a (minimum one) ES provider v) if and only if the ES provider secures ES provision (conditionality). This definition has been criticized for being too narrow and thus excluding many payment schemes that do not comply with these criteria; in particular, the voluntary aspect of the transaction has been questioned at least from the buyer’s side since many PES cases rather involve governmental intervention and public payment schemes (Schomers and Matzdorf, 2013). PES remain a controversial topic and entail many challenges, such as accurately measuring how changes in agricultural practises will change ES delivery (Pascual et al., 2015). Perhaps most importantly, PES must be socially equitable to not overly favour large landowners as well as both economically efficient and environmentally effective (Baveye et al., 2016; Pascual et al., 2015; Schomers and Matzdorf, 2013). See Schomers and Matzdorf (2013) and Engel et al. (2008) for comprehensive reviews of PES and their application in developed and industrialising countries.

5 Land management and planning

This section broadens the scope of the information presented so far to discuss the implications for situational application and general soil management and monitoring, detrimental effects on soil resulting from soil contamination and the possibility of green infrastructure and nature-based solutions to improve soil quality and provide ecosystem services.

There seems to be a general consensus that **when a land (or soil quality) management strategy incorporates the concept of ecosystem services, quantifiable soil features can be more easily linked to land-use expectations and protection goals in a defensible and transparent way** (Bünemann et al., 2018; Burges et al., 2018, 2016; Epelde et al., 2014a, 2014b; Faber et al., 2013; Faber and Van Wensem, 2012; Garbisu et al., 2011; Gómez-Sagasti et al., 2012; Gutiérrez et al., 2015; Pulleman et al., 2012; Rutgers et al., 2012). Faber and Van Wensem (2012) argue that, while cost-efficiency may still be an important objective for the current state of soil protection, the land use perspective and 'suitability (or fitness) for use' are playing increasingly important roles. According to the 'fitness for use' principle, the level of soil quality required in a given location or site depends on the specific end use envisaged for such site; thus, a soil that has an excellent quality for one purpose can have poor quality for another (Gómez-Sagasti et al., 2012). Consequently, the choice for references in soil quality or risk assessment is critical, in which we have to accept that soil communities change as a result of land use, and strive for multi-functionality in soil based on sustainable management of functional soil biodiversity to enhance the provision of ES (Faber and Van Wensem, 2012). In the case of contaminated sites, given the possible risks posed by contaminants, polluted soils cannot be treated as isolated entities because a certain degree of contaminant dispersion and ecosystem interconnectedness is likely, which should elicit be caution when applying the 'fitness for use' principle (Gómez-Sagasti et al., 2012). However, as stated by Volchko et al. (2019a), '*contaminated sites, which are often found in attractive locations of a city map, should be managed in accordance with the soils' capability and their best condition.'*

The multi-functionality of soils is reflected as a basic principle of the modern understanding of soils, and soil functions are effectively introduced into spatial planning to assess the ecological value of soils and also to evaluate the loss of functionality caused by soil degradation (Lehmann and Stahr, 2010). The end use of a site will also have a direct impact on the targeted soil functions, and the end use of the site, in turn, will also impact the soil services resulting from the functions (i.e. demand for ES will differ depending on land use) (Lehmann and Stahr, 2010; Volchko et al., 2013). Broadly speaking, integrating soil functions and ecosystem services into land management and planning can occur on many different scales (i.e. micro- to macro-scale). One prominent example for doing so comes from Lehmann and Stahr (2010) who proposed a hierarchical concept for 'planner-oriented soil evaluation' divided into three levels, see Table 5-1. A general approach (level 1) would be appropriate for macro-scale planning on the international, national or city level which should consider soil function broadly, as specified in Soil Strategy (EC, 2006); an intermediate approach (level 2) would entail accounting for soil sub-functions (e.g. via soil quality indicators) when comparing land use alternatives for meso-scale planning of zones, city districts, etc; and a detailed, more site-specific approach (level 3) would be necessary for micro-scale planning when aiming to improve or optimise soil functioning for a specific type of land use (e.g. water regulation for stormwater management) (Lehmann and Stahr, 2010).

Table 5-1. Proposed hierarchy of soil functions and planning levels to respect both the multi-functionality of soils and the land-use specified soil evaluation, adapted from (Lehmann and Stahr, 2010).

Levels of eco-functions	Types of functions	Example application	Planning levels
General - Level 1	Functions and part-functions	Soil as a basis for life and living space for wildlife	Principal planning
Intermediary - Level 2	Land-use oriented sub-functions	Soils under extensive meadows and forests and infiltration bodies in high mountainous areas	Comparison of land-use alternatives
Detailed - Level 3	Soil performances from special relevance for spatial planning	Drainage for alternative stormwater management	Improvement of a fixed type of soil use

With very few exceptions, all soil organisms are ultimately driven by energy which is derived from reduced forms of carbon, and the C transfer with associated energy flows is the main integrating factor in ecosystem functioning (Brussaard, 2013; Kibblewhite et al., 2008). This implies that manipulations of the soil biota, induced by the living plants, plant litter, and soil organic matter, would affect ecosystem functioning with possible feedback to aboveground biota (Brussaard, 2013). Furthermore, the soil and plant microbiome (i.e. all microorganisms present in soil, rhizosphere and plants) fulfil crucial roles in ecosystem functioning like nutrient cycling, plant nutrient uptake and disease suppression, which ultimately regulates plant health, physiology and performance (Schröder et al., 2018). In particular, the biogeochemical cycles and biodiversity in soil, shown in Figure 5-1, have been highlighted as key drivers of ecosystem services provided by soils, which can influenced, positively or negatively, by land and soil management activities (Smith et al., 2015). Various aspects of soil management and best practices will briefly discussed in the following section, for as stated by Smith et al. (2015), '*enough is known* [*about the relationships between different facets of soils and the array of ecosystem services they underpin*] *to implement best practices now. There is a tendency among soil scientists to dwell on complexity and knowledge gaps rather than to focus on what we do know and how this knowledge can be put to use to improve the delivery of ecosystem services.*'

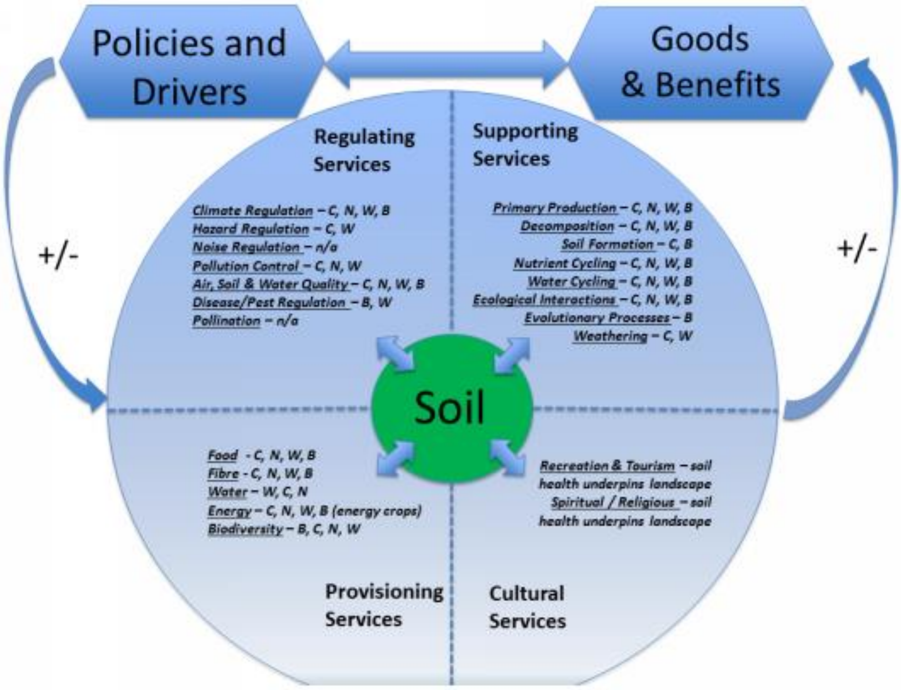


Figure 5-1. Schematic representation of where soil carbon, nutrient and water cycles, and soil biota underpin ecosystem services. Role in underpinning each ecosystem service shown by C, soil carbon; N, soil nutrients; W, soil water; and B, soil biota. Only soil carbon, nutrient, and water cycles, and soil biota are considered, so the figure does not represent a comprehensive overview of soil ecosystem services, from (Smith et al., 2015).

5.1 Soil management

This section is not intended to provide a comprehensive overview of soil management theory and practices, which has been covered extensively in other studies and reviews (see e.g. (Bai et al., 2018; Barrios, 2007; Brussaard, 2013; FAO et al., 2020; Kibblewhite et al., 2008; Orgiazzi et al., 2016; Turbé et al., 2010; Wall et al., 2012)), but instead aims to provide a few key aspects to consider and a selection of best practices.

Kibblewhite et al. (2008) maintain that soil is the site of a vital range of ecosystem functions which provide humans with a range of essential services. In natural ecosystems, these functions and services are driven by the energy generated by carbon transformations carried out by the soil biological community acting in a highly interactive and integrated fashion. Their study is carried out from the perspective of soil health for sustainable agriculture systems, which must ensure that the full range of ecosystem services is conserved for future generations by retaining the multi-functional capacity agricultural soils and is heavily impacted by humans. The authors list four main factors controlling soil health (quality) that must be accounted for in soil management:

1. **Soil type** – considering past land management by humans and how the natural soil has been altered, for example by loss of surface horizons due to erosion, alteration of soil water regime via artificial drainage, salinization due to poor irrigation practices, loss of natural soil organic matter caused by arable production or contamination. Variable factors include physico-chemical parameters such as pH, bulk density, soil organic matter content nutrient availability and concentrations of toxic materials determine the overall condition of the soil

system and biological factors like the presence or absence of specific assemblages and types of organisms.

2. **Organisms and functions** – accounting for biodiversity within and between trophic groups and assemblages, including functional diversity and redundancy.
3. **Carbon and energy** – considering the available carbon, ultimately derived from net primary productivity, that is the energy driving soil systems (the 'common currency'). This suggests that the quantity and quality of organic matter pools may be indicative of the state of the soil system, while the flows and allocations of carbon between assemblages or organisms may provide information about their relationships to ecosystem functions.
4. **Nutrients** – considering nutrients as a controlling input to the soil system and the processes within it. Their levels and transformations are critical to soil health (quality). After carbon, the cycling of nitrogen and phosphorus to, from and within the soil system most affects its dynamics and the delivery of ecosystem services, including agricultural production. Manipulation of nutrient supplies to increase productive outputs from the soil system by the addition of fertilizers has been one of the keystones of agriculture for centuries. Nonetheless, knowledge is limited about the impacts of nutrient additions on the condition of different assemblages of soil organisms and thence on their functions.

Kibblewhite et al. (2008) further discuss the significant impact of agricultural interventions (e.g. the use of pesticides, powered tillage and the use of inorganic sources of nutrients) upon the biological communities of soil, which can damage their habitats and disrupt their functions to varying extents. They claim that the link between disturbance, targeted biota and effect on function is far from linear owing to the high level of interaction between organisms and functions. Cause-effect relationships are complicated by a few major 'integrating features' of the soil community, including i) *energy flow* (i.e. .C transfers) – the majority of the soil organisms depend directly or indirectly via one or more trophic levels on the processes of organic matter decom-position for their source of energy and carbon, and any disruption of this energy generating system may result in changes in the flow of energy and carbon dedicated to the different functions; ii) *multi-functionality* – the probability of soil organisms participating in the ecological processes governing more than one function, which may mean that organisms belonging to a certain functional assemblage play a role in a separate function; iii) *soil as a habitat* – the activities of soil organisms are influenced by the condition of their habitat in the soil, but at the same time continuously modify it, and any shift in one function is thus likely to influence others by habitat change (Kibblewhite et al., 2008). In terms of management, the agricultural soil system is a subsystem of the 'agroecosystem', and the majority of its internal functions interact in a variety of ways across a range of spatial and temporal scales that can provide a framework for management options (Kibblewhite et al., 2008; Turbé et al., 2010). The assumption is that the size of organisms strongly determines their spatial aggregation patterns and dispersal distances, as well as their lifetimes, with smaller organisms acting at smaller spatio-temporal scales than larger ones (Turbé et al., 2010). Thus, according to Turbé et al. (2010), chemical engineers are typically influenced by local scale factors, ranging from micrometres to metres and short-term processes, ranging from seconds to minutes. Biological regulators and soil ecosystem engineers, on the other hand, are influenced essentially by factors acting at intermediate spatio-temporal scales, ranging from a few to several hundreds of metres and from days to years. Land managers have then two distinct management options for soil biodiversity: direct actions on the functional group concerned, or indirect actions at greater spatio-temporal scales than that of the functional group concerned (Turbé et al., 2010). For example, microbial activity is fundamentally governed by the availability of fixed carbon (the major ‘currency’ of the soil system), which is amenable to manipulation via agronomic factors such as crop type, and residue and other organic waste management (Kibblewhite et al., 2008). Organic matter plays other important roles in modulating soil functions, for example via the provision of surface charges, expressed as the cation exchange capacity, or influencing hydrological properties such as wettability (Kibblewhite et al., 2008). The intricacies of soil management for agroecosystem design and management are undoubtedly complex, so, to better illustrate these connections, Brussaard (2013) created a framework with a series of 'entry points' for management of important aspects of the soil system (Figure 5-2).

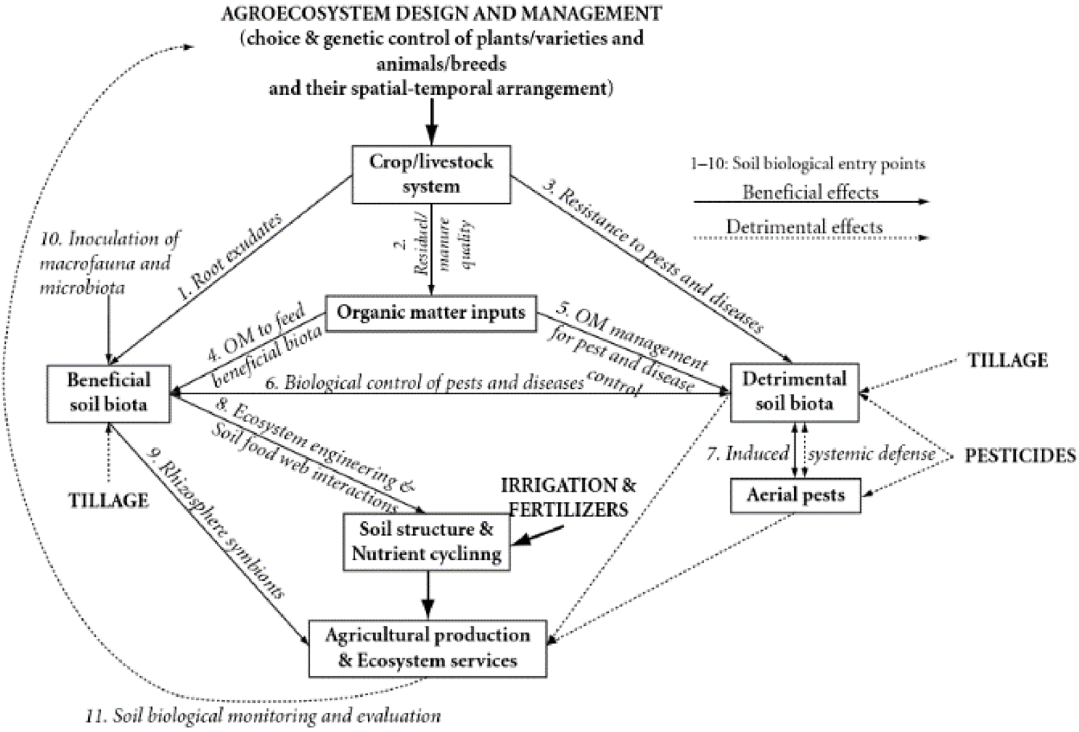


Figure 5-2. Agroecosystem design and management framework illustrating the potential entry points (1-10) for biological management of crop/livestock systems, organic matter inputs and soil organisms, aimed at sustainable agricultural production and ecosystem services, and feedback to agro-ecosystem design and management using monitoring and evaluation (11). OM = organic matter. From (Brussaard, 2013).

In terms of best practices for practical, sustainable management of agricultural soils to maximise soil health, the guide from Ward Labs (2019) lists five key components for soil health:

1. **Keeping the soil covered –** keeping the topsoil in place and resistant to erosion from wind, water, drying out in the sun and breakdown of soil aggregates.
2. **Minimising soil disturbance** – allowing and maintaining soil structure and aggregation by avoiding overgrazing and tillage and chemical over-use.
3. **Plant diversity –** diversity aboveground supports diversity belowground – polyculture instead of monoculture, preferably with perennial plants.
4. **Continual live plant/root –** carbon root exudates to soil microbiology during the entire growing season – Cover crops can address a number of resource concerns including: i) harvesting CO_2 and sunlight which provides carbon root exudates to the soil microbes; ii) building soil aggregates and pore spaces which improves water infiltration; iii)

covering the soil which controls wind and water erosion, soil temperature, and rainfall compaction; iv) weed suppression; wildlife food, habitat, and space; v) and pollinator food and habitat.

5. **Livestock integration** – incorporating livestock in the fall or winter can covert high carbon crop residue into low carbon organic matter, which balances the carbon to nitrogen ratio. Utilising livestock on annual or perennial plants in short bursts followed by long recovery periods allows plants to regrow and harvest sunlight and CO2. Integrating livestock will reduce nutrient export from cropland and hay fields, which will lead to the recycling of the majority of nutrients, minerals, vitamins, and carbon. Grazing manages weed pressure and reduces livestock waste associated with confinement which helps to manage our water quality and nutrient management concerns.

On a broader scale, the Landmark project[10] has developed a framework for 'functional land management', which is a conceptual framework for optimising the supply of soil-based ES, grouped into the five overarching soil functions of primary productivity, water purification and regulation, carbon sequestration and regulation, provision of functional and intrinsic biodiversity and provision and cycling of nutrients (see (O'Sullivan et al., 2015; Schulte et al., 2014)). A key result of the project has been the development of the Soil Navigator decision-support system (DSS) that is aimed at assessing and optimising the five above-mentioned soil functions by offering targeted solutions and management recommendations for farmers and farm advisors.[11] Another important result was clearly showing the expected trade-offs in soil function delivery for differing land uses and soil multi-functionality to deliver at least 3 soil functions at a site, as shown in Figure 5-3.

Figure 5-3. Illustration of typical suites of soil functions under contrasting land use types with colours corresponding to the five soil functions respectively, from (Schulte et al., 2014).

Turbé et al. (2010) also provided insight into the expected trade-offs in both soil functioning and soil biodiversity that could be expected with different land use types. They emphasised that grassland soils present the richest soil biodiversity, and it is worthwhile to consider including longer-lasting grasslands in an arable crop rotation in order to restore carbon levels and soil biodiversity, as well as disease-suppressing services. In Europe, grasslands generally host the

[10] Landmark 2020
[11] www.soilnavigator.eu

most diverse and abundant earthworm communities, with some dominated by endogeic species and others by anecic species but can be degraded by soil compaction and intensive agriculture. Depending on the land-use type, different representation of species within the three main functional groups can be expected, as shown in Table 5-2.

Table 5-2. Distribution of functional groups by land-use types with + or – indicating overall impact, adapted from (Turbé et al., 2010).

Soil biodiversity (Dominance/diversity)	Forest	Grassland	Cropland	Urban land
Total	++	++	+	-
Chemical engineers	Fungi dominated	Fungi dominated; Fungi: 10-100m; Bacteria: 10^8-10^9 g/soil	Bacteria dominated; Bacteria: 10^8-10^9 g/soil	Bacteria dominated
Biological regulators	Fungal-feeding protists and nematodes (100-1000 g/soil), Micro-arthropods (10^6/m²)	Protists and nematodes dominated; Protists: 1000/g; Nematodes: 10-100/g; Micro-arthropods: 5000-20000/m²	Opportunistic bacterial-feeding fauna; Protists: 1000/g; Nematodes: 10-20/g; Micro-arthropods: <100/m²	Negligible
Ecosystem engineers	Earthworm and ant-dominated; Anecic earthworms (100/m²)	Earthworm dominated; Endogeic/Anecic earthworms	Epigeic and endogeic earthworms (50-300/m²)	Negligible

According to Turbé et al. (2010), the key feature of agricultural land uses (i.e. croplands), is the specialisation of the production process, often resulting in monocultures and choice of fast-growth and high-yield plants that allocate most of their biomass to the harvested parts. The systemic ramifications of this type of land use are best stated by the authors (exact wording as in source, (Turbé et al., 2010) pp. 130):

> "In other words, conventional agriculture may push ecosystems in the direction of performing one single service, food provisioning, at the expense of the other, related services, such as the maintenance of soil structure, water quality and climate control. Such intensive agricultural practices contribute to the homogenisation of the landscape and are unfavourable to most soil organisms, leading to large scale soil biodiversity changes. It is not necessarily so that soil biodiversity of croplands is so much less than of for example grasslands, but some essential species groups with special functions can drop out. For example, cropland soil contains relatively few arbuscular mycorrhizal fungi and few earthworms. The soil community is adapted to regular disturbance and the food chains are mainly based on bacteria-based pathways. Especially conventionally cropped soils result in stressed and depleted soil food webs. When intensively cropped, arable soils are characterised by low organic matter inputs (leaf litter and stubbles are largely removed), and thus low soil fungal/bacterial ratios, and depleted bacteria-dominated chemical engineer communities. Consequently, biological regulator communities are themselves reduced and dominated by opportunistic bacterial-feeding fauna. Finally, strong mechanical and chemical disturbance cause reduction of earthworm and mycorrhizal fungi communities. Earthworms are only present at moderate densities (10 à 20 individuals per m²) and mostly composed of endogeic species, as epigeics are missing due to a lack of litter layer. Together, these conditions are indicative of low resilience and low sustainability" (Turbé et al., 2010).

Although each type of land use is characterised by its specific soil biodiversity, the intensity of management practices may also vary within a certain land use and severely impact soil biota (Turbé et al., 2010). According to the so-called 'intermediate disturbance hypothesis', soil biodiversity peaks at intermediate management intensities; whereby, species diversity and abundance increase from low to intermediate disturbance (e.g. extensive grasslands to organic agriculture), peak at moderate agricultural disturbance (e.g. organic agriculture) and then decrease with strong agricultural disturbances (e.g. intensive monocultural agriculture) (Turbé et al., 2010). Therefore, reducing management intensity of an intensive cropping practice with some degree of organic inputs, continuous plant cover and limited tillage, typically leads to an environment in which soil biodiversity is enhanced (Turbé et al., 2010). In addition to management practices, land-use changes can also have a positive or detrimental effect on soil biodiversity. For example, due to historic land-use changes and management, current communities are often composed of generalist species that have been able to adapt well to changes leading to a distinctly homogenous population of, for example, earthworms across Europe with local structural differences (Turbé et al., 2010). Homogenisation of biological communities poses challenges to resilience of communities to future changes, and ongoing changes (e.g. converting forests to grasslands or croplands) can also induce rapid changes in soil communities that may limit their ability to provide ecosystem services, see Table 5-3. For example, in the case of deforestation, stormwater run-off and the associated risk of erosion are increased with decreasing vegetation cover, and it has been observed that **land without vegetation can be eroded 123 times faster than land covered by vegetation which lost less than 0.1 ton of soil per ha/year** (Turbé et al., 2010).

Table 5-3. Impact of land-use change on the diversity of the three functional groups and the ecosystem services provided, adapted from (Turbé et al., 2010).

Functional group	**Forest → Grassland**	**Grassland → Cropland**	**Cropland → Urban land**	
Chemical engineers	⬂ - fungi; ⬀ - bacteria	⬂ (but some local ⬀)	⬂	
Biological regulators	= / ⬀ ⬀ - nematodes ⬂ - micro-arthropods	⬂ Plant-feeding → bacteria-feeding nematodes	⬂	
Ecosystem engineers	⬀ Anecic → endogeic earthworms	⬂ / 0 ⬂ - anecic earthworms	⬂	
Ecosystem service	**Forest → Grassland**	**Grassland → Cropland**	**Cropland → Urban land**	**Affected soil functions**
Soil fertility and nutrient cycling	= / ⬂	⬂	⬂	Reduced decomposition of SOM; Reduced biological control
Regulation of carbon flux and climate control	⬂	⬂	⬂	Reduced decomposition and mixing of SOM
Regulation of the water cycle	-	⬂	⬂	Reduced burrowing activity
Decontamination and bioremediation	-	⬂	⬂	Impaired self-regulation of ecosystems
Pest control	-	⬂	⬂	Reduced biological control
Human health effects	-	-	-	-

5.1.1 Adaptive soil management and monitoring

Much of what has been discussed thus far in this review has the ultimate purpose of being applied for use in soil quality assessment and monitoring (e.g. the use of reliable, relevant indicators connected to ecosystem services). A key consideration for this section is the incorporation of 'adaptive management and monitoring' (hereafter referred to as just 'adaptive monitoring') schemes into long-term land management and planning (see e.g. (Birgé et al., 2016; Chapman, 2012; Epelde et al., 2014a; Hooper et al., 2016)). The 'adaptive' in adaptive monitoring refers to the strategy of monitoring programs evolving iteratively (i.e. continuously improving or 'learning by doing') to reduce uncertainty regarding responses in soil biodiversity to management as new information emerges or as research questions or objectives change (Birgé et al., 2016; Chapman, 2012; Epelde et al., 2014a). Furthermore, adaptive monitoring should be firmly based on ecosystem services (e.g. as assessment endpoints), thereby providing the best means to develop necessary information for informed decision-making (Birgé et al., 2016; Chapman, 2012; Epelde et al., 2014a; Hooper et al., 2016). Common pitfalls in monitoring programs that can be avoided through adaptive monitoring include selecting the wrong drivers (e.g. politics rather than good science), poor initial design and lack of clarity regarding goals, components, data collection and communication (Chapman, 2012). Also, to be most effective, monitoring plans should be designed concurrently with restoration plan development and implementation and integrated early into the planning stage (i.e. in the beginning of the site assessment and management process) to support restoration goals for the site (Hooper et al., 2016; Hull et al., 2016). Birgé et al. (2016) provide an instructive example of an adaptive management framework tailored towards managing soil biodiversity to enhance ecosystem services, see Figure 5-4.

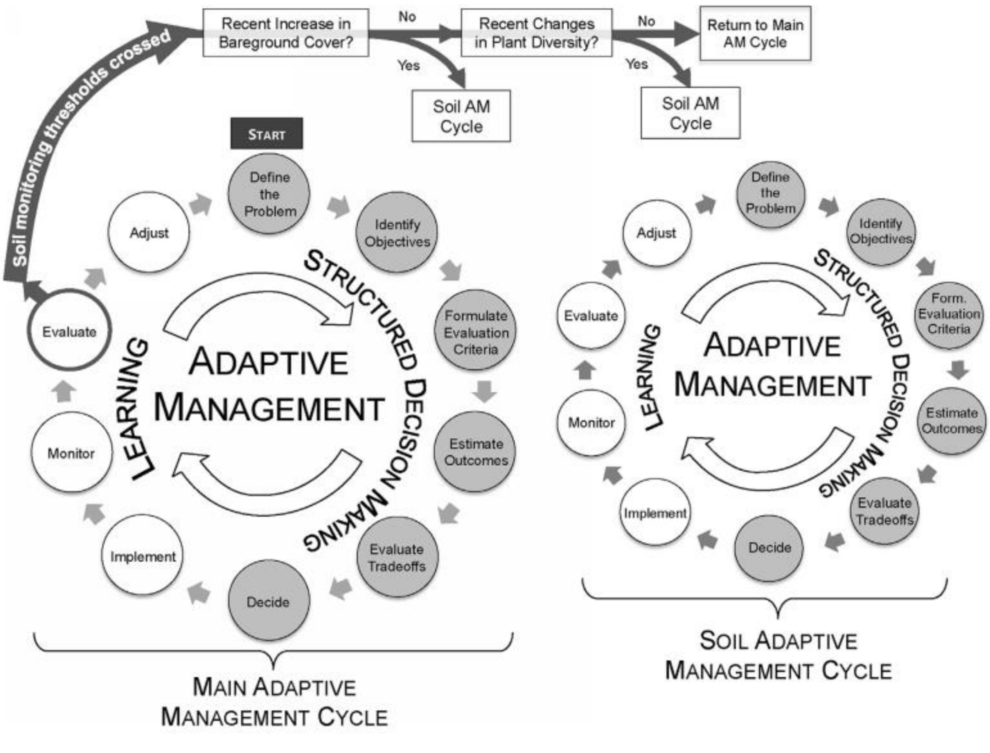


Figure 5-4. An adaptive management framework for reducing uncertainty in the soil system while proceeding with the fundamental management objective, from (Birgé et al., 2016).

Adaptive monitoring has also been recommended to monitor the remediation or restoration of soils at contaminated sites, both in terms of risk reduction and return of lost ecological function and ecosystem services (Epelde et al., 2014a; Hooper et al., 2016; Hull et al., 2016). For example, Epelde et al. (2014) proposed to incorporate adaptive monitoring into the long-term monitoring required for metal phytostabilisation at contaminated sites by including a few key principles, such as i) posing well-formulated, good questions that are based on testable hypotheses to cover the chemical, toxicological and ecological concerns related to contaminated soils; iii) basing the study on a conceptual model of the ecosystem under study; iii) avoiding the common mistake of including a long, expensive 'laundry list' of indicators and instead focus on analytical techniques that are cheap, easy to interpret and do not require specific expertise – preferably soil microbial parameters as indicators of soil health and recovery; iv) grouping the soil microbial parameters into higher-level categories such as 'ecological attributes' (vigour, organisation, stability) or ecosystem services in order to facilitate interpretation and stability over time against changes in techniques, method, interests, etc.; v) evolving the monitoring program iteratively (e.g. **every 5 years**) to critically analyse the monitoring program and remediation effectiveness. To facilitate this process, the authors created a 'Phytostabilisation Monitoring Card', see Figure 5-5, based on both ecological attributes and ecosystem services and using soil microbial parameters to answer the question 'Does the recovered soil health remain as such?' To evaluate, initial values can be normalised to 100% (100% = value obtained for each specific variable measured at the beginning of the long-term monitoring program) and the mean value = arithmetic mean of all microbial parameters included in each ecological attribute or ecosystem service (mean > 100% indicates a positive trend; mean value < 100% indicates a negative trend) (Epelde et al., 2014a).

QUESTION: Does the recovered soil health remain as such?	MICROBIAL PARAMETERS	MEAN VALUE
ECOLOGICAL ATTRIBUTES		
VIGOR	Microbial biomass C, basal and substrate-induced respiration, potentially mineralizable N	% Initial value
ORGANIZATION	Structural and functional diversity measurements via PCR-DGGE	% Initial value
STABILITY	Drought-stability assays with microbial biomass C, potentially mineralizable N and PCR-DGGE (structural and funcional)	% Initial value
ECOSYSTEM SERVICES		
RENEWAL, RETENTION AND DELIVERY OF NUTRIENTS FOR PLANTS	Potentially mineralizable N, soil enzyme activities (urease, acid and alkaline phosphatase, arylsulphatase)	% Initial value
HABITAT AND GENE POOL	Structural and functional diversity measurements via PCR-DGGE	% Initial value
DISPOSAL OF WASTES AND DEAD ORGANIC MATTER	Microbial biomass C, basal and substrate-induced respiration, potentially mineralizable N, soil enzyme activities (β-glucosidase, urease, acid and alkaline phosphatase, arylsulphatase)	% Initial value

Figure 5-5. Phytostabilisation Monitoring Card, based on both ecological attributes and ecosystem services, for soil microbial parameters as selected to answer the question 'Does the recovered soil health remain as such?', from (Epelde et al., 2014a).

5.2 Contaminated sites and marginal land

In urban soils, the conditions for soil biodiversity are severely stressful as they are often substantially degraded or altered by the impacts of urbanisation (Pavao-Zuckerman, 2012, 2008). Nevertheless, there exist ecosystems in and around cities that can provide critically needed and highly demanded ecosystem services provided that the ecosystem functionality is retained or restored (see e.g. (Pavao-Zuckerman, 2012, 2008) for interesting discussions on urban ecology and ecological restoration in cities or (Craul and Craul, 2006) for *getting the soil* *right* in terms of landscape architecture and soil design). In fact, there have been many studies documenting the unique biodiversity present at contaminated sites or other marginal land that is worth protecting and preserving (FAO et al., 2020; Garbisu et al., 2020; Hartley et al., 2008; Orgiazzi et al., 2016; Turbé et al., 2010; US EPA, 2009). However, in many cases, the soil in urban areas is sealed (i.e. paved over with hard, impermeable surfaces), which stops all exchange between soil fauna and all external inputs, prompting chemical engineers to go into a dormant, inactive state in the sealed soils thereby terminating provided services (Turbé et al., 2010). If they do not simply die off, profound shifts in community structure can also be expected in non-sealed soils such as domination by bacteria as chemical engineers given the high chemical inputs used for pest control, biological regulators being dominated by microarthropods and earthworms being mostly absent or present only in urban parks or forests (Turbé et al., 2010).

Soil contamination is a major and acute problem in many areas of the EU, which has received growing attention in recent years (Panagos et al., 2013; Turbé et al., 2010). In order to capture the full ecological value that these sites offer, tools such as SF Box will be required to address the question: 'what can this soil actually do and can it perform its functions well, assuming that it is free of contaminants?' (Volchko et al., 2019). Applying such tools and asking these kinds of questions will assist in making a distinction between the effects of contamination on soil biota and the effects of soil capability (i.e. soil quality) to function as a host to these species in its own reference state free of contaminants (Volchko et al., 2019). With regards to the effects of contaminants on soil biota, the impacts of chemical pollution on soils can be extremely heterogeneous, acting either directly (e.g. mortality, inhibition of growth or reproduction) or indirectly (e.g. altering community structure and food webs or inhibiting necessary functions) on specific organisms and trophic levels depending on such factors as the type of contaminant, mode of action, distribution in the soil matrix and concentration (Turbé et al., 2010). Turbé et al. (2010) provide extensive discussion regarding the effects of various contaminants on the three main functional groups, summarised in Table 5-4. The authors conclude that organic matter degradation and soil structure regulation are the functions most impacted by contamination which can impair the delivery of services like nutrient cycling, soil fertility and water control. Important considerations for each functional group include the following (Turbé et al., 2010):

- **Chemical engineers** – chemicals can have differing impacts different soil microbial species and communities, which can disturb the interactions within and among functional groups. Also, due to their very short reproduction time (e.g. an average of 20 minutes for bacteria in optimal conditions), exposure to some toxic chemical could rapidly lead to a resistant population which can develop genes that can be transferred to successive generations and even aid in breaking down some organic contaminants into non- or less toxic compounds.

- **Biological regulators** – many chemicals have deleterious impacts on biological regulators by affecting lifespans and reproduction, the most studied being nematodes. However, dose-response effects can vary depending on the chemicals and exposure time.

- **Ecosystem engineers –** earthworms, in contrast to ants and termites which tend to be more resistant, are often highly sensitive to contaminants due to their close contact with pore water and their highly permeable epidermis (easily absorbing water-soluble contaminants) and the fact that they ingest large quantities of soil. However, sensitivity to contaminants varies on the earthworm species due to feeding habits and tolerance strategies (e.g. eliminate certain excess metals like Cu and Zn) as well as the bioavailability of the contaminant.

Table 5-4. Possible impacts of chemical pollution on soil biodiversity related services, on the basis of its impacts on soil organisms, adapted from (Turbé et al., 2010).

Chemical pollutant	Affected soil organisms	Affected soil function	Affected soil service
Pesticides	Biological regulators, ecosystem engineers	Organic matter decomposition, residue fragmentation	Nutrient cycling, Soil fertility
Pesticides	Chemical engineers (microorganisms), biological regulators (microfauna)	Mineralisation, immobilisation	Nutrient cycling, Soil fertility
Pesticides	Ecosystem engineers	Bioturbation, soil structure regulation, SOM production	Nutrient cycling, Soil fertility, Water regulation
Pesticides	Biological regulators	Population control	Pest control
GM plants	Chemical engineers	Mineralisation, organic matter decomposition	Nutrient cycling, Soil fertility
GM plants	Ecosystem engineers (earthworms)	Soil structure regulation, SOM production and transformation	Nutrient cycling, Soil fertility, Water regulation
Industrial chemicals and heavy metals	Chemical engineers	Mineralisation, organic matter decomposition	Nutrient cycling, Soil fertility
Industrial chemicals and heavy metals	Biological regulators (nematodes)	Soil structure regulation, SOM production and transformation, regulation predation	Nutrient cycling, Soil fertility, Pest control, Water control, Climate control
Industrial chemicals and heavy metals	Ecosystem engineers (earthworms)	Soil structure regulation, SOM production and transformation	Nutrient cycling, Soil fertility, Water regulation

As noted by Hartley et al. (2008), there is an assumption that if appropriated site conditions are provided then natural processes will 'take care of the rest', which is an ill-defined concept, but refers to the self-organising capacity of ecosystems to recover by assembling the necessary constituents for improved soil functioning. With respect to contaminated soils, gentle remediation (e.g. bioremediation, phytoremediation or using soil amendments) of the toxic contaminant to an environmentally safe, non-toxic level could be applied as a soil management strategy to ameliorate soil conditions and promote ecological recovery (Burges et al., 2018; Cundy et al., 2016; FAO et al., 2020; Gómez-Sagasti et al., 2012; Orgiazzi et al., 2016; Thomsen et al., 2012; Turbé et al., 2010). For instance, soil fauna ('ecosystem engineers') can also be dispersal agents for both microorganisms that degrade organic contaminants and the contaminants themselves through the soil profile (FAO et al., 2020). Soil invertebrates such as earthworms have also been shown to improve decontamination of organic (e.g. pesticides) and inorganic contaminants (metals) by plants and microorganisms (FAO et al., 2020; G. Lacalle et al., 2020; Orgiazzi et al., 2016; Rodriguez-Campos et al., 2014; Turbé et al., 2010). Natural decontamination processes or bioremediation are even regarded as an 'regulating ecosystem service' performed by microorganisms, earthworms and other soil organisms functioning in healthy soils; therefore, a high diversity and biological activity within soils, especially at the level of chemical engineers, but also in the case of ecosystem engineers, is indispensable to ensure this essential service (FAO et al., 2020; Orgiazzi et al., 2016; Turbé et al., 2010).

A good example of this type of land management strategy was shown by Schindelbeck et al. (2008) for a contaminated urban vacant lot; where, the poor soil condition (low organic matter and nutrient content) could be ameliorated through the addition of organic soil amendments to restore soil quality and also provided a soil cap as a barrier to reduce risks posed by elevated metal concentrations in the soil (e.g. direct ingestion or spreading via dust). Along these same lines, Schröder et al. (2018) envisioned a strategy for 'mobilising' marginal lands to intensify production and utilise these otherwise latent resources for biomass production and improved economic, environmental and social outcomes, shown as a decision tree in Figure 5-6.

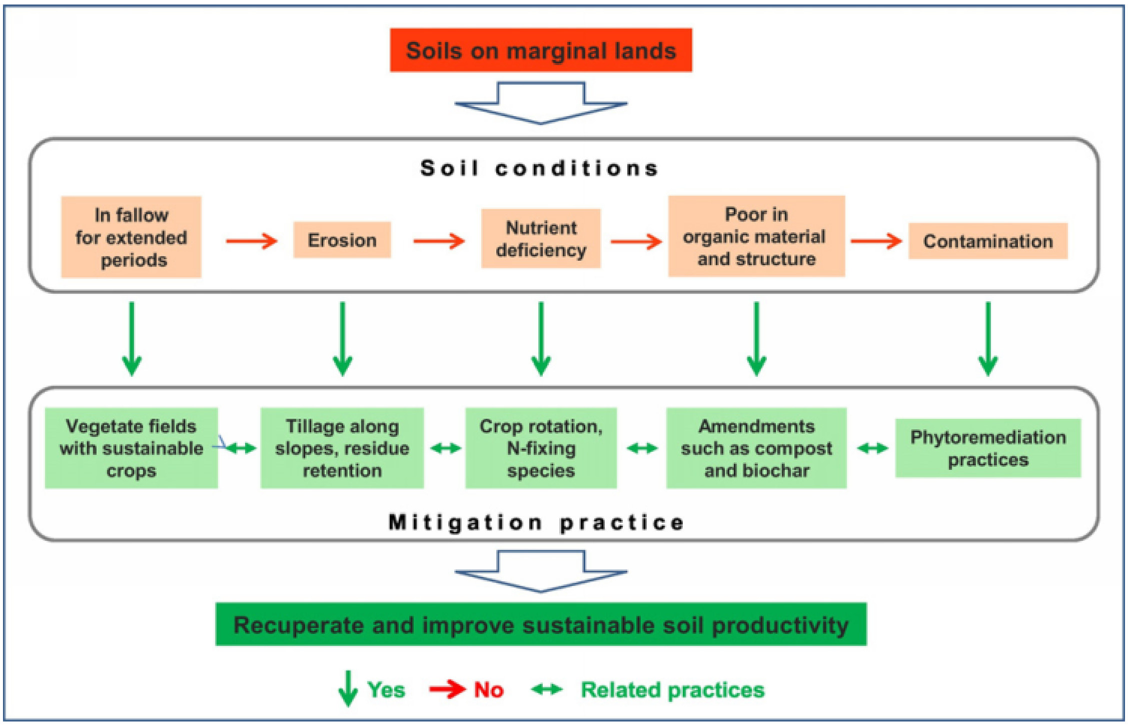


Figure 5-6. Decision tree for improving and optimising the productivity of soils on marginal lands, from (Schröder et al., 2018).

Schröder et al. (2018) emphasise that in order to achieve this vision, there must be a focus on integrated, systems-based approaches of land management with sustainable intensification of agricultural production, even on neglected or marginal sites: underexploited grassland, abandoned and set aside lands and brownfields with actual or aged pollution. On these marginal sites, the land has lost its economic and/or ecological viability for the community and typically is degraded to such an extent that it has lost the capacity to provide ecosystem services (Schröder et al., 2018). Approaches to restore the functionality of such sites should be based in on 'eco-agricultural' (or regenerative agriculture) practices that entail applying organic matter in the form of crop residues and other wastes or compost or in the later years also biochar, to enhance biogeochemical nutrient cycling, stimulate soil biodiversity and its proliferation effectively (Schröder et al., 2018). Soil amendments play a key role in this strategy to improve soil fertility by using organic amendments like compost and biochar to e.g. adjust the soil pH, increase soil nutrient content and retention capacity and improve the microbial community abundance and activity (Schröder et al., 2018; Touceda-González et al., 2017b, 2017a) (see Appendix IV for a comprehensive table synthesising the relevant properties of compost, animal manure, digestate and biochar from (Schröder et al., 2018)).

There is a broad range of possible soft re-use strategies for brownfields that can provide a host of benefits either as a permanent or interim measure, which have been extensively covered in projects like the Holistic Management of Brownfield Regeneration[12] (HOMBRE) (Bardos et al., 2016). A major outcome of the Hombre project was the 'Brownfield Opportunity Matrix' that can be used to identify opportunities for additional services and benefits that can be delivered by brownfield projects via soft end-uses to increase their overall value and

[12] Brownfield Regeneration: Holistic Management - FP7 HOMBRE Project (zerobrownfields.eu)

attractiveness for investment (Bardos et al., 2016). Soft re-uses are mostly mediated by plants, and brownfields that are significantly covered by tree, shrub and grass vegetation contribute to the variety of urban green space as vital elements of 'urban green infrastructure' and host particular ecosystems that provide important ecosystem services like habitat and biodiversity, biomass production and micro-climate regulation (Bardos et al., 2016; De Valck et al., 2019; Mathey et al., 2018, 2015; Pueffel et al., 2018). Urban green brownfields can even provide highly valued cultural ecosystem services such as recreational services for local residents that can serve as alternatives to and providing similar functions as more classic green spaces such as parks and gardens (Mathey et al., 2018, 2015; Pueffel et al., 2018).

6 Discussion and concluding remarks

The provision of soil-based ecosystem services is clearly dependent upon soil ecosystem functioning, which is the result of interactions between soil biota and their physic-chemical environment, and there is a need for at least a minimum appreciation of soil quality in contaminated site assessment to ensure the provisioning of these crucial services. Nowadays, as shown in Figure 6-1, soil quality assessment has developed to focus on the multi-functionality of soils, providing ecosystem services and resistance and resilience to disturbances. In terms of applying the concepts covered in this review to soil quality and ecosystem service assessment at contaminated sites, some of the main findings are the following:

Soil quality and ecosystem service assessment must entail clear **objectives** that are decided upon at the beginning of a site assessment and planning project; whereby, target ecosystem services and desired functionality are made explicit (in consultation with stakeholders) as assessment endpoints with which to assess the effectiveness of a restoration/remediation strategy.

The **selection of soil quality indicators** should be carried out according to a multi-criteria analysis (e.g. logical sieve) in which preference is placed on those that are reasonably inexpensive, easy to understand, accessible in laboratories and standardised as well as meaningfully based on linkages between indicators, soil functions and ecosystem services. The frameworks offered by ecological risk assessments could be highly useful; for example, to establish parallel lines of evidence (e.g. physical, chemical and biological) along increasing tiers of complexity with which to assess the ecological status using measurement endpoints linked to ecosystem services. Semi-quantitative ecosystem service mapping procedures could also be highly useful to demonstrate the value of urban green brownfields.

The **interpretation** of the data obtained from the indicators should be well-defined and ideally based on quantitative, statistical evaluation. Scoring curves, target values or reference values can be utilised to make sense of the indicators and aid in both interpretation and **communication**. The (dis)agreement of results obtained from different lines of evidence can be reconciled using mathematical procedures employed in ecological assessment. Similar to the integrated risk value used in ecological risk assessment, an aggregated **soil quality index** is often desired but is likely to be more useful when assessed in relation to specific soil functions or ecosystem services by grouping indicators accordingly.

Incorporation of soil quality indicators in **monitoring** programs to assess restoration/remediation effectiveness should be based on the principles of adaptive monitoring to have a clear initial plan and iteratively improve it by revisiting the objectives approximately every 5 years.

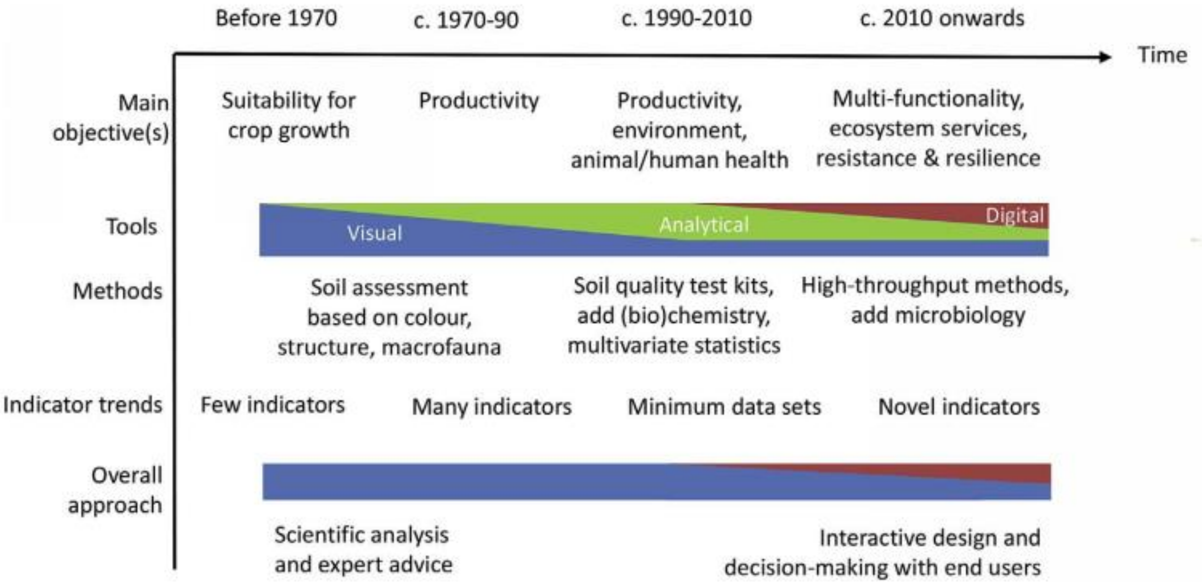


Figure 6-1. Main objectives, tools and approaches of soil quality assessment through history, from (Bünemann et al., 2018).

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Appendices

8 Appendix I

As of February 3, 2020, a blanket search for "ecosystem services" will result in 30, 791 hits and "soil functions" 1656 hits. The highest cited paper by Costanza et al. (1997) has been cited 11,057 times since its publication. Filtering through these results and using sensitive search terms to find those that are most relevant to this literature review is a challenge. Also, previously conducted reviews and seminal works (primary sources) have been extracted and will be referred to specifically for reference throughout this review. Additionally, certain papers have been recommended by professionals in the field so accordingly extra weight will be placed upon them. The table below shows the process of finding relevant literature in the Scopus database in this phase of the literature review:

Search Terms – Phytoremediation/GRO Smorgasbord

Feb. 3, 2020

"Phytoremediation" AND "_____"

Search Terms	***Hits***	***Year of Origin***	***Highest Citation Score***	***Relevance***
AND "meta-analysis"	18	2007	Audet and Charest 2007 (94)	8
AND "systematic review"	8	2017	Wang et al. 2017 (90)	2
LIMIT to *reviews*	986	1996	Haritash and Kaushik 2009 (1468)	
LIMIT to *reviews* + *heavy metal*	218	1997	Ali et al. 2013 (1133)	
LIMIT to *phytoextraction*	933			
LIMIT to *degradation*	421			
AND "removal rates"	300			
AND "soil quality" OR "soil health" OR "soil fertility"	381	1995	Garbisu et al. and Epelde et al. papers	
LIMIT to keyword - *soil quality*	157	2001		10
AND "risk management"	33	2000	Kuppusamy et al. 2017 (146)	7
Feb. 4, 2020				

AND "ecosystem services"	71	2008	Dickinson et al. 2009 (169)	22
"phytotechnology"	133	2000	Rezania et al. 2016 (141)	
+ "pollution" or "contamination"	61	2003	-	6
+ "remediation"	38	2002	-	6
"phytomanagement"*	171	2005	Robinson et al. 2009 (147)	15
LIMIT to *reviews*	10			5
+ "soil quality" or "soil health" or "soil fertility"	17			8
Feb. 7, 2020				
AND "plant selection" (screening)	48	1997	Labeau et al. 2008 (218)	-
AND "bioaugmentation"	177		Kuiper et al. 2004	-

*Usage of the term "phytomanagement" varies per paper, e.g. 'phytomanagement' can mean: 1) Greenland/Robinson et al. definition of maximizing co-benefits or 2) Phytoremediation to 'manage' a site using vegetation

Search Terms – Pre-conditions

Feb. 4, 2020

"Phytoremediation" AND "_____"

Search Terms	***Hits***	***Year of Origin***	***Highest Citation Score***	***Relevance***
AND "site-specific"	58	1998	Mulligan et al. 2001 (956)	7
AND "site characterization"	12			1
AND "site suitability"	0			
AND "site conditions"	24			1
AND "pre-conditions"	0			

AND "conditions"	2615		Haritash et al. 2009 (1468)	
AND "brownfield"	40	1999	Mench. Et al. 2010 (217)	16
AND "uncertainty"	41	1998	Mench et al. 2010 (217)	9
AND "regulation"	483			
And "monitoring"	799			
Feb 13, 2020				
AND "technosol"	33	2013	Sylvain et al. 2016 (41)	-
Feb. 14, 2020				
AND "indicators"	340	1994	He et al. 2005 (693) – Garbisu et al., Epelde et al.	-
September 3, 2020				
AND bioaccessibility	21	2006	Mench et al. 2006 (88)	6 (8)
AND bioavailability	941	1995 - Salt et al.	Haritash et al. 2009 (1584)	

Search Terms – Soil functions

Feb. 4, 2020

"Phytoremediation" AND "_____"

Search Terms	*Hits*	*Year of Origin*	*Highest Citation Score*	*Relevance*
And "ecosystem services"	71	2008	Dickinson et al. 2009 (169)	22
AND "soil functions"	14	2006	Gomez-Sagasti et al. 2012 (88)	8

"Brownfields" OR "contaminated sites" (or "contaminated land" or "marginal land" or "polluted soil" or "contaminated land" or "polluted land) AND "_____"

Search Terms	***Hits***	***Year of Origin***	***Highest Citation Score***	***Relevance***
AND "soil functions"	35	2002	Van Straalen 2002 (69)	10
AND "soil quality"	230	1994	Luo et al. 2012 (306)	-
AND "soil health"	31	2000	Dickinson et al. 2009 (169)	8
AND "ecosystem services"	73	2003	Dickinson et al. 2009 (169)	
AND "ecosystem services analysis"	0			
AND "ecosystem services mapping"	1	2018	Cortinovis and Geneletti 2018 (9)	1
AND "ecosystem services assessment"	0			
AND "green infrastructure"	20	2011	Mathey et al. 2015 (31)	-
AND "nature-based solutions"	7	2016	Song et al. 2019 (22)	3
Feb. 6, 2020				
AND "ecological risk assessment"	149		Linkov et al. 2009 (143)	-
LIMIT to "ecosystem services"	3	2012	Thomsen et al. 2012 (39)	3
March 24, 2020				
AND "decision support"	203		Li et al. 2007 (153)	-

August 21, 2020

"Brownfields" OR "contaminated sites" (or "contaminated land" or "marginal land" or "polluted soil" or "contaminated land" or "polluted land) AND "____"

Search Terms	***Hits***	***Year of Origin***	***Highest Citation Score***	***Relevance***

AND "minimum data set"	2	2014	Volchko et al. 2014 (30)	2
AND "soil quality indicators"	14	1999	Schindelbeck et al. 2008 (62)	6
AND "soil functions	56		Hinojosa et al. 2004 (175) (Gomez-Sagasti et al. 2012 (88))	
AND "soil health"	78		Dickinson et al. 2009 (178)	
Oct 10, 2020				
And "ecosystem service mapping"	5	2014	Gret-Regamy et al. 2014 (40)	2
AND "ecosystem services"	196			
Oct 20, 2020				
And "ecosystem service assesment"	0			
And "ecosystem service valuation"	0			
And "ecosystem service analysis"	1		Wells et al. 2018 (6) - marginal agricultural lands	

Search terms – Ecosystem services

Feb 12, 2020

Search Terms	*Hits*	*Year of Origin*	*Highest Citation Score (of relevance)*	*Relevance*
"ecosystem services mapping"	128			
AND "ecological risk assessment"	87	Sergeant, A. 2000 (6)	E.g. Faber 2013	14
"soil ecosystem services"	167		Dominati et al. 2010 (362)	

March 5 - *soil ES*				
And "endpoints"	109	Cairns Jr. 1994 (21)	Keeler et al. 2012 (201) + Faber et al.	-
AND "indicators"	2643		Lavalle et al. 2006 (664)	-
AND "typology"	230			
And "demand"	2200			-
And "future needs"	34			-
And "design"	2250			-
And "optimize"	389			-
And "semi-quantitative"	27	Everard et al. 2009 (3)	Schipanski et al. 2014 (157)	2
And "remediation"	266		Becerril et al. Garbisu et al., Mench et al., Cundy et al., Epelde et al.	
April 20 , 2020				
"soil ecosystem health"	33	Lau et al. 1997 (11)	Thomsen et al. 2012 (41) Park et al. 2011 (38) - nematodes Chae et al. (13) - beta-glucosidase	3

9 Appendix II

Compilation of connections between soil organisms and soil functions and ecosystem services, from Bünemann et al. (2018) – Supplementary material (see paper for references).

Soil organism	Main soil functions	Mechanisms involved	Soil-based ecosystem services	Ease of application	References
Macroorganisms (fauna)					
Earthworms (macrofauna)	Soil structure maintenance, decomposition, organic matter and water cycling, habitat provision	Burrowing, fragmentation of litter, soil aggregation, humification, organic matter distribution	Biomass production, erosion control, water supply, climate regulation, biodiversity conservation	Easy to sample but not ubiquitous	Blouin et al. 2013, Lavelle et al. 2006
Nematodes (microfauna)	Element cycling, decomposition, biological population regulation	Grazing on microorganisms, root herbivory, predation	Biomass production, pest and disease control	Identification via morphology currently only by specialists but facilitated by molecular tools in the future. Ubiquitous, easy to sample, abundant, sensitive. Key role in soil food web. Information about feeding preferences and life strategy.	Mulder et al. 2005, Neher et al. 2001, Schloter et al. 2003
Protists (microfauna)	Element cycling, biological population regulation	Grazing on microorganisms	Biomass production	Poorly defined taxonomically, difficult to isolate and identify. Variable in space and time.	Foissner 1999, Riches et al. 2013

Collembola (mesofauna)	Decomposition, element cycling, biological population regulation	Grazing on fungi	Biomass production, pest and disease control	Cumbersome to sample and isolate, difficult to identify	Brussaard et al. 2004, Cardos et al. 2013, Pulleman et al. 2012, Ruf et al. 2003
Enchytraeids (mesofauna)	Decomposition, soil structure maintenance	Burrowing, fragmentation of litter, soil aggregation, decomposition, humification, organic matter distribution	Water supply, climate regulation	Easy to sample but difficult to identify	
Mites (mesofauna)	Decomposition, element cycling, biological population regulation	Grazing on bacteria and fungi, fragmentation of residues	Biomass production, pest and disease control	Cumbersome to sample and isolate, difficult to identify	
Macroarthropods (macrofauna)	Soil structure maintenance, biological population regulation	Burrowing, root herbivory, predation, grazing on bacteria and fungi	Biomass production, pest and disease control, biodiversity conservation	Relatively easy to sample, taxonomically very diverse	
Microorganisms (microbes)					
Bacteria	Element and organic matter cycling, decomposition, biological population regulation	Symbiotic association (nitrogen fixing bacteria), production of antibiotics, transformation and mineralization of organic material	Biomass production, pest and disease control, climate regulation	Spatially and temporally variable. Taxonomically very diverse and difficult to classify.	Barrios 2007, Lehman et al. 2015, Schloter et al. 2017

Fungi	Element, organic matter and water cycling, soil structure maintenance, decomposition, biological population regulation	Symbiotic association (mycorrhizae), production of antibiotics, transformation and mineralization of organic material	Biomass production, water quality and supply, erosion control, pest and disease control, climate regulation		

Examples of soil organisms, their main soil functions and ecosystem services, and gaps and opportunities, from FAO et al. (2020).

Table 3.1 | Examples of organisms, their main soil functions and ecosystem services, and gaps and opportunities

Organisms	Main Soil Ecosystem Functions	Ecosystem Services	Gaps and Opportunities
Megafauna (e.g. moles, beavers, armadillos)	***Bioturbators***: soil, organic matter and sediment redistribution to greater depths	***Supporting services***: *Soil formation* *Nutrient cycling* ***Regulating services***: *Climate regulation* *Disease and pest regulation* *Water regulation (water availability, including regulating extremes – drought and flood)* *Remediation* *Pollination* ***Provisioning services***: *Food* *Freshwater* *Fuel* *Fibre* *Biochemicals* *Genetic resources* ***Cultural services***: *Spiritual, recreational, symbolic values of landscapes*	***Lack of data and limited knowledge on***: Organisms and their functional roles Interaction between organisms and their communities, as well as with terrestrial biodiversity (including crops) Distribution of soil organisms globally Better understanding of how biodiversity loss by anthropogenic activities might affect stability of food webs and ecosystem functioning Better understanding of the impact of climate change on soil communities Improvement of biological control practices for pests and pathogens ***Opportunities***: Undescribed microorganisms may have a wide range of metabolic capabilities influencing ecosystem services Application of soil microorganisms for specific industrial, agricultural or environmental purposes
Macrofauna (e.g. earthworms, ants, termites, millipedes, insects)	***Ecosystem engineers***: fragment, rip, and tear organic matter, providing smaller pieces for decay by organisms; mineralization of organic matter; ***Bioturbators***: moving and mixing soil, increasing water permeability and soil aeration		
	Bioremediation: *Eisenia fetida* earthworms can accumulate cadmium and some other pollutants like polycyclic aromatic compounds (PAHs)		
Mesofauna (e.g mites, springtails)	***Soil modifiers***: all mesofauna modify the fine structure thereby changing many soil microhabitat features. ***Decomposers***: micro and mesofauna feed on dead plant material or living microbes to recycle nutrients for primary production. Thereby they modify the fine-scaled structures of soil. ***Food web stabilizers***: micro and mesofauna operate as predator and/or prey to regulate and shape soil biological communities.		
Microfauna (e.g. protozoa, nematodes)			
Microbes (e.g. virus, bacteria and archaea, fungi)	***Bioremediation***: break down, removing, altering, immobilizing, or detoxifying various chemicals and physical wastes from the environment like PAHs (see chapter 5) ***Decomposers***: weathering minerals; Carbon transformation by decomposition of organic matter and storage, nutrient cycling by transforming inorganic compounds into forms usable by plants, regulate soil structure and pathogens		
	Gas producers: methanogenic archaea transfer of C, N_2, N_2O, CH_4 denitrification		
	Nitrogen fixers: rhizobia bacteria found on legume roots helping to increase nitrogen uptake.		

10 Appendix III

Indicators used in ecosystem service mapping procedure, from Ivarsson (2015).

Indicators for quantifying changes in provision of urban ecosystem services:

	Ecosystem service	Indicator	Reference
Provisioning	Food	1) Production/Harvest (ton/year), 2) Areas available suitable for production (m^2)	Hauser et al. 2011, Mazza L., et al. 2011, Baggethun et al. 2013, Egoh et al 2012
	Fresh water	Groundwater generated (m^3/ha/year;m^3/year)	Hauser et al. 2011, Baggethun et al. 2013
Regulation & Maintenance	Air quality regulation	Leaf area index (Area of vegetation (ha))*	Burkhard, B. et al. 2012, Egoh, B et al 2012
	Climate regulation global	1) Carbon bound in ecosystems = C sequestration (ton C/year; ton C/ha/year), 2)O_2-CO_2 balance (+/- kg C /year) Production or reduction of other GHG (kg/yr; kg/ha/year)*	Mazza L., et al. 2011
	Climate regulation local (urban climate)	Area of vegetation (ha)	Nowak et al 2012
	Water regulation	1) Area of vegetation (ha), 2) water storage capacity (m^3/ha/year)	1)Egoh, B et al 2012 2)Maes J. et al 2011
	Noise reduction	1) Leaf area (m^2) and distance to roads (m): [dB(A)]/vegetation unit (m)* 2) Area of vegetation (ha)	1) Baggethun et al. 2013 2)Maes J. et al 2013
	Water purification and waste treatment	Volumes of soil available for filtration (m^3/ha)	Baggethun et al. 2013
	Pollination and seed dispersal	1) Species diversity and abundance of birds and bumble bees* 2) Area of vegetation (ha)	1)Baggethun et al. 2013 2) Egoh, B et al 2012
	Maintaining nursery populations and habitats	Conservation status of habitats and species; number of species for which the GI element provides habitat*.	Mazza L., et al. 2011
	Natural hazard regulation	Natural water retention capacities (m^3)*	COWI 2014
Cultural	Knowledge systems	Participation, reification and external source of social-ecological memory.*	Baggethun et al. 2013
	Aesthetic values	Scenic landscape (ha) (e.g. revealed through prices on real estate)	COWI 2014
	Cultural heritage values	Number of visitors/tourists	
	Recreation and ecotourism	1) Area of green public space (ha)/inhabitant (or every 1000 inhabitant) 2) Number of visitors/tourists	Baggethun et al. 2013

*Area of vegetation (ha) is used as proxy for the suggested indicator.

Indicators for quantifying changes in the provision of soil ecosystem services:

Ecosystem service		Soil context	Indicator	Reference
Provisioning	Food	Nutrient cycling to support plant growth (primary production) including food and fiber production	1) Production/Harvest (ton/year), 2)Areas available suitable for production (m^2)	Hauser et al. 2011, Mazza L., et al. 2011, Baggethun et al. 2013, Egoh et al 2012
	Biomass	Basis of all terrestrial ecosystems –life support	Soil (horizon) development and disturbance regime controls ecosystem development.*	Finvers 2008
Regulation & Maintenance	Fresh water	Water purification and soil contaminant reduction. Contaminants are adsorbed into soil aggregates, by clay particles and organic matter, and degraded, (chemically altered) by soil biota	Volumes of soil available for filtration (m^3/ha)*	Finvers 2008
	Climate regulation global	Carbon sequestration and Regulation of greenhouse gasses	1) Carbon bound in ecosystems = C sequestration (ton C/year; ton C/ha/year)*, 2) O2-CO2 balance (+/- kg C /year) Production or reduction of other GHG (kg/yr; kg/ha/year)	Mazza L., et al. 2011
	Water regulation Erosion regulation	Flood regulation	Natural water retention capacity (m^3)*	COWI 2014
	Water purification and waste treatment	Remediation of soil contaminated by diffuse airborne pollution. Soil biota metabolize contaminants through oxidative or reductive processes.	1) Nutrients levels (C,N,P), moisture (40-60 % of field capacity), appropriate pH (~7) and T (15-45 C), oxygen for oxidative process*. 2) Removed/immobilized pollutants (kg N, P, C, heavy metals, pesticides, and other pollutants (y^{-1})	1)Finvers 2008 2)COWI 2014

*Area of vegetation (ha) is used as proxy for the suggested indicator.

The ecosystem services including in the mapping procedure in Applicera, from Volchko et al. (Volchko et al., 2020).

Ecosystem service: CICES class	SEPA subclass	1	2	3: How does this ES create a benefit?	4	5: P	5: F
Bioremediation by microorganisms, algae, plants, and animals	Bioremediation by plants	ML	H, M*	The subareas are covered by tree species *Salix* which are used for phytoremediation. This species naturally occurs at the site and most likely stabilises metals in the upper soil layer.	L	M	M
	Bioremediation by microorganisms	ML	H, M*, L1	The possibility that soil microorganisms at the site take part in the transformation of organic substances should not be excluded.	L	M	M
Regulation of chemical composition of atmosphere and oceans	Global climate regulation	Y	H, M*, L1	The greenery at the site sequesters carbon dioxide and contributes to the regulation of the concentrations of gases in the atmosphere that influence the global climate.	S	H	H
Regulation of temperature and humidity, including ventilation and transpiration	Regulation of local and regional climate	Y	H, M*, L1	The greenery contributes to the regulation of physical air quality for people, and increases thermal comfort.	L	M	M
Noise attenuation	Noise attenuation	Y	L1	The trees in subarea L1 reduce the stressful impact of noise on people.	L	M	M
Hydrological cycle and water flow regulation (Including flood control, and coastal protection)	Dampening of run-off and floods	Y	H, M*, L1	*Birch* species present at the site are known to be good at absorbing water. In general, the greenery at the site has the potential to dampen run-off and prevent floods.	S	M	M
Control of erosion rates	Erosion control	Y	H, M*, L1	The greenery at the site prevents soil loss.	S	H	H
Buffering and attenuation of mass movement	Landslide risk reduction	Y	L1	The trees at the site stabilise the slope and reduce landslide risks.	S	H	H
Pollination (or 'gamete' dispersal in a marine context)	Pollination	Y	H, M*	*Salix* species are pollinated by honey bees and bumble bees. This maintains the abundance of the *Salix* species, which also provide other services at the site.	L	M	M
Maintaining nursery populations and habitats (Including gene pool protection)	Habitat for nursery populations	Y	H, M*	*Salix* species present at the site bloom in the early spring, which makes them critical for the survival of newly hatched honey bees and bumble bees. Sustainable populations of useful species contribute to additional services (yield of fruit crops) in other ecosystems.	S	M	M

(1) "Is ES present at the site?" – N: No; ML: Most likely; Y: Yes. (2) "What areas are important for this ES?" – H: Subarea H; M*: Subarea M*, L1: Subarea L1 (see Fig. 2). (3) "How does this ES create a benefit?". (4) "Who benefits from this ES?" – L: local stakeholders; S: society. (5) "Is there a demand for this ES?" – P: at present; F: in the future; A: Absent; M: Medium demand; H: High demand. CICES is the Common International Classification of Ecosystem Services (CICES, 2018). SEPA is the Swedish Environmental Protection Agency (SEPA, 2012).

11 Appendix IV

Compilation of organic amendments and their relevant properties, from Schröder et al. (Schröder et al., 2018). Green and orange colour indicates positive and negative effects respectively, yellow colour indicates presence of both positive and negative effects, grey colour indicates lack of knowledge. Numbers indicate specific references, see (Schröder et al., 2018) for sources.

Properties	COMPOST[1]	ANIMAL MANURE[2]	DIGESTATE (anaerobic digestion)[3]	BIOCHAR[4]
Increase in content of organic matter	increases soil organic matter, humic substances	increases soil organic matter, depends on animal diet	depends on feedstock - humic acids (mainly solid fraction)	affects the stability of existing organic matter
Modification of C:N ratio			low C/N ratio due to digestion	increase
Improvement of water holding capacity	Increases		improves	increases due to surface structure
Supply of nutrients (N, P, etc.) nutrient balance	enhances nutrient supply	leaching of N and P – content differs with animal species	depends on feedstock - mineral N, P (mainly liquid fraction), possible leaching	reduces leaching of nutrients / slow release fertilizer - provides P and K
Modify pH	lowers pH		high pH	increase in soil pH of acidic soils
Modification of cation exchange capacity	Increases			increase in soils with low CEC
Improvement of texture and aggregation state	amelioration of structure and porosity	reduces density	reduces density, increase in aggregate stability	increase in porosity, stability of aggregates
Sequestration of pollutants/contaminants	through humic substances		not reported	can sequester pollutants, but also increase mobility
Addition of pollutants/contaminants	might contain persistent pollutants	micronutrients supplied to animals	might contain persistent pollutants, metals	can contain pollutants, in this case it is not usable
Decrease in salinity	Improvement		can increase salinity with repeated applications	can sequester salts and modify CEC
Soil conservation (e.g. minimise erosion)	remediates degraded soils		still to be investigated	still to be investigated
Increase in microbial biomass	increase	Increase	considerable increase	increase
Increase in microbial diversity	increase or decrease	Increase	significant changes	significant differences
Stimulation of specific microorganisms	no indication	antibiotic resistance	dominance of slowly growing microorganisms	arbuscular and ectomycorrhiza
Increase in enzymatic activities	increase in soil microbial activity	Increase	nitrogen mineralization, other enzymes	reports on increase in enzymatic activities
Increase in diversity of fauna	Limited observations, differing effects		limited observation, increase	Limited observations, differing effects
Effects on plants growth	positive	very positive	positive	mostly positive
Increase of yield	Positive	Positive	fertilizer capacity	reports on increase of crop yield
Increase of product quality	not significant			not assessed
Improve in defense against pathogens	Positive effects			Limited observations, positive effects
Origin, raw materials	biomass from different sources		biomass from different sources	biomass from different sources
Production requirements	requires large amounts of energy, long time			depends on biomass feedstock - importance of temperature
Standardisation of product	Quality assessment differs in the countries	not possible	not possible	just starting
Cost (including transport)	moderate		depends on feedstock	depends on feedstock - high
Positive carbon emission	emissions during composting	emissions of CH_4 and N_2O, NH_3	during digestion GHG emissions, NH_3 emission	could stimulate CO_2 emissions by microbes
Negative carbon emission	carbon sequestration in humic substances		decrease of emissions from manure	removal during growth of biomass, C - sequestration
Legislation, norms on applicability	Differences among countries		can be amendment or fertilizer	limited
Social acceptability	well established	well established	Low	not yet tested
Additional benefits (e.g. energy production)	scalable to farm		production of biogas	reduction of N_2O emissions
Ecosystem services of relevance				